

WATER ENCYCLOPEDIA



Surface and Agricultural Water

Edited by
JAY H. LEHR
JACK KEELEY

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WATER ENCYCLOPEDIA

SURFACE AND AGRICULTURAL WATER

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PREFACE

Surface water and agricultural water are uniquely associated as they provide many of our basic needs, including food and fiber, power, transportation, and recreation. Like other volumes in the *Water Encyclopedia*, we have selected articles as varied in content as they are in technical sophistication. To this end, the reader will also recognize that single topics are occasionally duplicated at varying levels of scientific acumen.

Articles are also provided that demonstrate that surface and agricultural water are associated in yet another way: They must be used efficiently and protected to assure their productiveness far into the future. For example, agricultural water use efficiency is discussed from several viewpoints with respect to irrigation technology. River basin planning is approached in diverse ways, including stream classifications, watershed hydrology, modeling, erosion control, and water conservation.

We have necessarily included articles addressing issues of quality with respect to both surface and agricultural water. In addition to an assessment of pollution outflow from agricultural areas, the quality of reclaimed irrigation is addressed from both chemical and microbial standpoints. Watershed areas are examined according to their contribution and vulnerability to contamination, flooding, sediment transport, and trace elements.

Discourses on surface water would not be complete without articles related to fish. Accordingly, we have included articles on fish growth, fisheries, fishponds, and the use of fish scales in toxicological studies as examples.

Another vital area of study in this volume is perhaps best described as the practical side. These areas are of less esoteric origins, including salt tolerance of plants, irrigation wells, weed control, tile drains, and moisture content in to agriculture. Similar topics in surface water include riparian systems, reservoir design, wetlands, lakes, levees, and the unit hydrograph.

Finally, and *appropriately*, *this volume of the Water Encyclopedia* contains articles on specific water bodies and the consequences of their being. Included are the Aral Sea, the Ganga River of India, the Great Lakes, and the Yellow River in China, only to name a few. Here, too, the association of surface water and agricultural water are reinforced. This volume presents an important segment of the topic of water. We believe that the reader's educational pursuits will be well met by its contents.

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SURFACE WATER HYDROLOGY

ACIDIFICATION—CHRONIC

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The objective here is to describe and discuss in general terms, processes that acidify freshwaters (including lakes, rivers, and streams), agents of acidification, watershed features associated with sensitivity to acidification, and associated chemical and biological effects. This is a complicated topic. A variety of acidifying agents and a number of physical and chemical watershed characteristics can make a particular body of water susceptible to acidification. Chronic, long-term acidification of water is addressed, not the short-term (episodic) acidification that sometimes accompanies rainstorms or snowmelt.

First, it is necessary to define a few terms. *Acidification* is generally a decrease in the acid-neutralizing capacity (ANC) of water. It can also be defined as a decrease in pH. ANC refers to the capacity of a solution to neutralize strong acids. It can be measured in a laboratory, usually by the Gran titration procedure. It can also be defined in different ways, based on the measured values of various ion concentrations in the water. Many mathematical models of acid–base chemistry define ANC as the sum of the base cation concentrations ($\text{Ca}^{2+} + \text{Mg}^{2+} + \text{Na}^+ + \text{K}^+ + \text{NH}_4^+$ [termed SBC]) minus the sum of the mineral or strong acid anion concentrations ($\text{SO}_4^{2-} + \text{NO}_3^- + \text{Cl}^-$ [termed SAA]).

$$\text{ANC} = \text{SBC} - \text{SAA} \quad (1)$$

where all ions are in units of microequivalents per litre ($\mu\text{eq/L}$). ANC defined in this way is approximately equal to laboratory measurements (Gran titrated) of ANC only if the solution contains relatively low concentrations of dissolved organic carbon (DOC) and Al. Both of these latter constituents cause Gran ANC to differ from ANC defined as $\text{SBC} - \text{SAA}$ (1). ANC reflects the extent to which added strong acids can be neutralized, or buffered, by the nontoxic base cations in solution such as Ca^{2+} rather than by H^+ and Al^{3+} , which are toxic to some species of aquatic animals and other life forms. The term *acidic* is used to describe a lake, river, or stream (surface water) that has ANC below zero. In other words, if the water is acidic, then the sum of the concentrations of the strong acid anions exceeds the sum of the concentrations of the base cations ($\text{SAA} > \text{SBC}$). Thus, a body of water can become *acidified* by increasing the concentration of one or more of the SAA components, by decreasing the concentration of one or more of the SBC components, or a combination of both. There is a rather consistent relationship between pH and ANC, although varying levels of DOC contribute to scatter in this relationship. At $\text{ANC} = 0$, the pH is generally near 5 (Fig. 1).

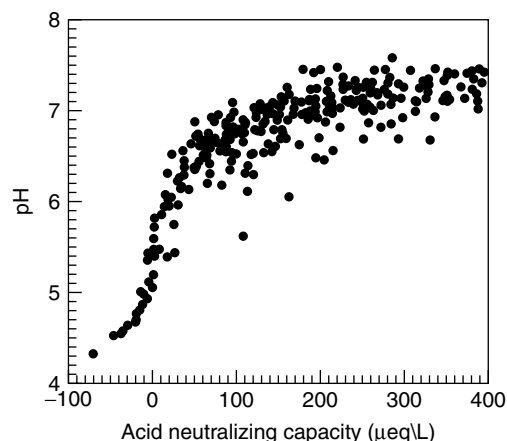
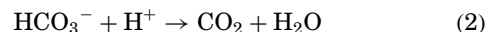


Figure 1. Relationship between pH and ANC in streams sampled by the U.S. EPA's Environmental Monitoring and Assessment Project (EMAP) in the Mid-Atlantic Appalachian Mountains.

Weak acids in solution, such as carbon dioxide, aluminum hydroxides, and organic acids, act as buffer systems that prevent dramatic changes in water pH upon adding small amounts of strong acid. When CO_2 from the atmosphere dissolves in water, it forms carbonic acid (H_2CO_3), which readily dissociates into hydrogen and bicarbonate (HCO_3^-) ions. Bicarbonate constitutes the most important buffer system in most freshwaters because it undergoes the following reaction upon adding strong acid:



At low pH (<5.5), the bicarbonate buffering system becomes less important, and the aluminum hydroxide buffering system becomes proportionately more important. In waters that have high DOC, the organic acid buffering system can dominate. In addition to providing pH buffering upon adding strong acid, organic acids also reduce the pH of the solution by up to a full pH unit or more.

Chronic acidification of fresh waters in North America is usually associated with an increased concentration of SO_4^{2-} . The source of this SO_4^{2-} can be air pollution (by acidic deposition, also called acid rain) or geologic sources of sulfur. Acidic deposition is the process whereby S and/or N is transported in the atmosphere and deposited downwind of pollution sources onto water bodies and the terrestrial watersheds that drain into them. The major sources of atmospheric sulfur include coal-fired power plants and industrial facilities. Geologic S can be contributed to surface waters by acid mine drainage (AMD; 2) or other watershed disturbance and subsequent oxidation of sulfur-bearing minerals.

There are also examples of surface waters that have been acidified partially, or even mostly, by increased concentration of NO_3^- . In such cases, the source of the

NO_3^- is generally atmospheric. Nitrogen, in the form of nitrogen oxides and ammonia (which can be converted to NO_3^-), is emitted to the atmosphere largely from motor vehicles, industrial sources, and agriculture. NO_3^- is more likely to be a major contributor to short-term episodic acidification of surface waters, whereas SO_4^{2-} is more commonly associated with long-term chronic acidification. There are numerous exceptions to these general patterns, however.

It is also possible for waters to become chronically acidified due to Cl^- contamination, but this rarely occurs to any degree. Atmospheric deposition of Cl^- in sea spray can cause episodic acidification of surface waters in near-coastal areas, and road salt application can probably contribute in some cases to both chronic and episodic acidification, but there is little evidence that this is of any regional importance.

Thus, SO_4^{2-} , NO_3^- , and Cl^- , alone or in combination, can contribute to chronic acidification of surface waters. Where chronic acidification of North American waters occurs, it is most often from SO_4^{2-} .

Acid mine drainage is an important cause of sulfur-based chronic surface water acidification in mining regions of North America. This problem is especially prevalent in the northern Appalachian Mountains of West Virginia and Pennsylvania, where almost 10% of the streams are chronically acidic as a consequence of AMD (2). AMD results from the exposure to air and water of mine spoils containing sulfide minerals. The minerals are oxidized by a series of chemical reactions and microbial processes to yield high concentrations of sulfate and heavy metals in drainage water and low pH.

In many cases, the added SO_4^{2-} that contributes to chronic acidification is of atmospheric origin. However, it is important to note that, in most regions that receive high levels of S deposition, the primary determinant of whether, and to what extent, a body of water acidifies is not the amount of SO_4^{2-} that is added from the atmosphere and/or from geologic sources. Rather, the most important factor governing surface water acidification is usually the inherent sensitivity of the watershed. In most areas of North America, most watersheds are not sensitive to surface water acidification from atmospheric S deposition because abundant supplies of base cations in watershed soils usually exist. In such nonsensitive watersheds, increased contributions of SO_4^{2-} from the atmosphere are balanced by increased release of base cations from soils, and the drainage water does not acidify. However, some fraction of the surface waters in some regions are acid-sensitive, and increased concentrations of SO_4^{2-} can decrease surface water ANC. To understand this concept, it is helpful to consider the principle of electroneutrality.

The sum of the charges of all positively charged ions (cations) must equal the sum of the charges of all negatively charged ions (anions) in solution, so that

$$\sum \text{Cations} = \sum \text{Anions} \quad (3)$$

In relatively undisturbed freshwaters, these sums can be approximated as

$$\sum \text{Cations} = \text{SBC} + \text{H}^+ + \text{Al}^{n+} \quad (4)$$

and

$$\sum \text{Anions} = \text{SAA} + \text{HCO}_3^- + \text{Org}^- \quad (5)$$

where all ions are in units of $\mu\text{eq/L}$ and Org^- represents a mixture of ill-defined organic acid anions. The valence of Al in Eq. 4 is expressed as $n+$ because inorganic ionic Al in solution can include a number of species, with varying charges. At a pH less than about 4.5, much of the Al in solution is present as Al^{3+} . At somewhat higher pH (about 4.5 to 5.5), proportionately more of the Al in solution is present as Al hydroxides and Al fluorides whose charges are $+1$ or $+2$. At a pH above 5.5, there is little Al in solution. Other ions, such as F^- , Fe^{n+} , Mn^{n+} , for example, are usually present in minor amounts in natural waters and have little impact on the charge balance.

If the concentration of one of the SAA anions, for example, SO_4^{2-} , increases in solution, then one of the other anions (e.g., HCO_3^- or Org^-) has to decrease in concentration and/or one or more of the cations has to increase in concentration to maintain the electroneutrality indicated in Eq. 3.

Combining Eqs. 3–5, we can express the charge balance as

$$\text{SBC} + \text{H}^+ + \text{Al}^{n+} = \text{SAA} + \text{HCO}_3^- + \text{Org}^- \quad (6)$$

Equation 6 can be rearranged to yield the following:

$$\text{SBC} - \text{SAA} = \text{HCO}_3^- + \text{Org}^- - \text{H}^+ - \text{Al}^{n+} \quad (7)$$

and therefore,

$$\text{ANC} = \text{HCO}_3^- + \text{Org}^- - \text{H}^+ - \text{Al}^{n+} \quad (8)$$

Thus, ANC can be defined in more than one way. If Equation 1 holds true, then Equation 8 must also hold true to satisfy the electroneutrality constraint (Eq. 3).

These equations and definitions presume that the mixture of organic acid anions in the water (Org^-) is made up entirely of weak acid anions that contribute ANC to solution. However, Driscoll et al. (3) showed that about one-third of the organic acid anions in Adirondack Mountain, New York, lakes actually exhibit strong acid characteristics and behave more like SO_4^{2-} and other strong mineral acid anions. The remainder of the organic acids act as weak acids. Thus, the contribution of organic acid anions to surface water acid–base chemistry is rather complicated. These substances contain a multitude of functional groups, some of which contribute ANC and some of which lower ANC.

As SAA concentration in surface water increases, the concentration of one or more of the other anions must decrease, and/or one of the cations must increase in concentration to maintain electroneutrality. For surface waters that are relatively high in ANC (greater than about 50 to 100 $\mu\text{eq/L}$), most of the compensating change is usually an increase in the concentration of Ca^{2+} and other base cations (SBC). In such cases, there is little or no water acidification from the increase in SAA; both the SAA and SBC increase, but the $\text{ANC} (= \text{SBC} - \text{SAA})$ stays about the same. For surface waters that are relatively low

in ANC, in contrast, the compensating change generally involves multiple responses, including decreased HCO_3^- and increased SBC, H^+ , and Al^{n+} . Increased H^+ and Al^{n+} can be toxic to aquatic life. In some cases, especially if DOC is relatively high, Org^- also decreases. This latter effect can influence the relative distribution of Al forms between nontoxic organic complexes and inorganic species, some of which are toxic.

The effects of acidification on aquatic life are due mainly to the changes that occur in HCO_3^- , H^+ , and/or Al^{n+} . Any change in SBC to compensate for an increase in SAA does not result in acidifying water, although it can contribute to acidification of the soil. Because of the fundamental importance of these processes to water acidification, Henriksen (4) defined a factor, termed the F-factor, as the proportional change in SBC, as opposed to HCO_3^- , H^+ , and Al^{n+} , relative to the change in SAA:

$$F = \frac{\Delta \text{SBC}}{\Delta \text{SAA}} \quad (9)$$

In watersheds that are not acid-sensitive, $F \approx 1.0$. In highly to slightly acid-sensitive Adirondack lake watersheds, F varies from about 0.4 to near 1.0.

Thus, for every 1 $\mu\text{eq/L}$ increase in SO_4^{2-} concentration in an acid-sensitive lake or stream, the change in base cation concentrations will often balance much of that increase, and the resulting ANC change will generally be a rather small fraction (often less than half) of 1 $\mu\text{eq/L}$. There are, however, many highly sensitive watersheds in which ANC change constitutes an appreciable component of the increase in SAA. Within North America, such watersheds are especially prevalent in high mountainous areas of the western United States.

Waters sensitive to chronic acidification from acidic deposition generally had ANC less than about 50 $\mu\text{eq/L}$ prior to the advent of the acidic deposition. In some cases, surface waters having somewhat higher initial ANC (~ 50 to 100 $\mu\text{eq/L}$) have become acidified. Sensitive waters are usually found at moderate to high elevation, in areas of high relief (mountainous), rapid runoff of precipitation (flashy hydrology), and minimal opportunity for contact between drainage water and soils or geologic materials that may contribute weathering products (base cations) to solution. Sensitive streams generally occur in small watersheds ($< 10 \text{ km}^2$). These are generally the highest, smallest, coldest, and highest gradient streams and their associated lakes. Most acid-sensitive surface waters are underlain by bedrock that is resistant to weathering and/or contains only thin deposits of glacial till. Soils in the watersheds of acid-sensitive surface waters contain relatively small amounts of exchangeable base cations, and the soil base saturation is generally less than 10 to 15%. It is rare to find large lakes or large rivers that are acid-sensitive because large lakes and rivers have large watersheds and therefore, the probability increases that some of the bedrock and soils encountered by drainage water will be rich in base cations.

Chronic acidification of lakes and streams has occurred in areas throughout portions of the eastern United States, eastern Canada, and in parts of the upper Midwest.

The areas most heavily impacted include the Adirondack and Catskill Mountains in New York, portions of the Appalachian Plateau in West Virginia, southeastern Canada, the mountains of western Virginia, and the Upper Peninsula of Michigan (5). In some portions of these regions, as many as 10 to 50% of the lake and/or stream resources may have experienced some degree of chronic acidification from sulfur deposition. Some of the most acid-sensitive watersheds in the world occur in high mountain areas of the western United States, including the Front Range of Colorado, northern Rocky Mountains, Sierra Nevada, and Cascade Mountains. However, because the levels of acidic deposition are generally low in most parts of the western United States, the amount of actual chronic acidification has been rather limited in most of these highly sensitive areas. Aquatic ecosystems in the Front Range of Colorado have probably been the most impacted of these western mountain ranges to date, and there the acidification has been more strongly from N than from S.

In general, most of the SO_4^{2-} that is deposited from the atmosphere to a watershed acts as a mobile anion and moves through soils into surface waters. In other words, SO_4^{2-} inputs to the watershed in the form of wet, dry, and cloud deposition are approximately equal to SO_4^{2-} outputs in drainage water. This steady-state condition is approximated in most areas that were previously glaciated, for example, throughout the northeastern United States and southeastern Canada. In the southeastern United States, however, which was not glaciated, soils tend to be much older and more highly weathered. Such soils adsorb some of the deposited sulfur before it reaches surface waters. This process effectively prevents water acidification, at least temporarily. Over time, however, the capacity of the soils to adsorb S becomes depleted as the S adsorption sites on the soil become saturated by continuing S inputs. Therefore, proportionately more of the SO_4^{2-} inputs reach surface waters, potentially contributing to acidification. At the present time, the soils in many areas in the southeastern United States, generally from Virginia to Tennessee and Georgia, are gradually losing their capacity to adsorb S. This will have a profound impact on watershed responses to future changes in S deposition. Model projections suggest that even if S deposition is reduced by more than 70% from 1995 levels, most streams in this region will continue to show an increase in streamwater SO_4^{2-} concentration, which will contribute to further acidification (6). Thus, southeastern U.S. watersheds exhibit a delayed response to S deposition due partly to gradual changes in the extent to which S is adsorbed on watershed soils.

Base cation depletion is another important process that is believed to contribute to a delayed watershed response to acidic deposition. This process is not confined to a particular region, but rather can be important in any watershed that has a low base cation supply and receives moderate to high levels of acidic deposition. Because ANC reflects the difference between the concentrations of SBC and SAA, a decrease in base cation concentrations can contribute to acidification just as an increase in one or

more of the SAA constituents such as SO_4^{2-} or NO_3^- . It is believed that continuing levels of acidic deposition are depleting the exchangeable base cation stores in some watershed soils to the point that drainage water acidification is occurring in response to decreased base cation leaching from soils to drainage water. Such a process is not likely unless the soil base saturation (percent of the cation exchange capacity provided by base cations, rather than H or Al) is quite low, perhaps less than about 10%. Several factors can contribute to soils that have low base saturation, including long periods of acidic deposition, land use, (especially logging), and other disturbances that cause substantial erosion. In addition, some soils are naturally low in base cations as a consequence of the geologic makeup of their parent material and soil-forming processes. Recent decreases in the atmospheric deposition of base cations has also contributed to this problem.

In regions characterized by abundant wetlands, natural organic acidity from the breakdown of plant material can have a large impact on the acid–base chemistry of drainage waters. Lakes and streams in such regions are often characterized by relatively high concentrations of DOC (greater than about 5 or 6 mg C/L or 400–500 μM) and associated organic acids. Such lakes are often naturally acidic or low in ANC and show relatively little response of pH to changes in acidic deposition. For example, of the 560 Adirondack Mountain lakes considered most sensitive to acidic deposition (the thin till drainage and mounded seepage lake types), 45% had DOC higher than 6 mg C/L (500 μM) (7). Much of the acidity in these lakes is organic.

Chronic acidification can adversely impact many species of aquatic life, from algae to fish and fish-eating birds. Different species vary in their acid-sensitivity, as do different life forms within a given species. In lakes and streams of the eastern United States, much of the attention has focused on damage to native brook trout (*Salvelinus fontinalis*), which often would be expected in acid-sensitive lakes and streams. However, brook trout are relatively tolerant of acidity in comparison with some of the lesser known forage fish species such as blacknose dace (*Rhinichthys atratulus*). In most cases, brook trout are unable to live in waters that have chronic ANC less than about zero. Adverse episodic effects can occur at higher levels of chronic ANC in the range of 0 to 20 $\mu\text{eq/L}$ or higher. In general, the eggs and larval stages of fish are more acid-sensitive than adults.

Acidification of streamwater also results in adverse impacts on several important orders of aquatic insects, especially mayflies. Acidified streams tend to contain fewer species or genera of these insects than streams that have not acidified. In lakes acidified by acidic deposition, several species of *Daphnia* (Cladocera) are often highly impacted.

Many species of algae, especially diatoms, are acid-sensitive. In fact, the occurrence and relative abundance of the various diatom species can be used to infer past lakewater chemistry. This technique has been used to estimate the acid–base characteristics of lakes during preindustrial times, as a way of quantifying how much acidification has occurred in response to a century of acidic deposition. Such paleoecological studies have involved

collecting cores of lake sediment. The cores are then sliced into wafers, which are dated using radioisotopic dating techniques such as ^{210}Pb analysis. The remains of diatom species in each layer provide information about the lake conditions at the time that the sediment layer was deposited at the bottom of the lake (8). Such studies have shown, for example, that of the currently acidic Adirondack lakes, the median lake has acidified by about 37 $\mu\text{eq/L}$ since preindustrial times (9).

Current research on the chronic effects of surface water acidification is focused largely on documenting and predicting the extent of chemical and biological recovery that will occur in response to recent and projected future large decreases in atmospheric sulfur deposition. An additional important research focus concerns identification of the critical load of S or N deposition required to protect against surface water acidification to harmful levels or to allow recovery of acidified waters to chemical conditions that are no longer harmful to aquatic life. Much of this work is focused on lakes and streams in wilderness areas, national parks, and other protected areas.

Additional information on this topic can be found in books by Charles (5) and Sullivan (9).

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EPISODIC ACIDIFICATION

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Episodic acidification is the short-term decrease ($\leq 0 \mu\text{mol}_e \text{ L}^{-1}$) in acid neutralizing capacity (ANC) of surface waters due to both natural and anthropogenic factors. ANC is a measure of the ability of water or soil to neutralize added acids and is defined by the reaction of hydrogen ions with inorganic (e.g., HCO_3^-) or organic ions. Decreases in ANC generally result in a decline in pH, sometimes dropping to levels below 4.5 (1). Episodic acidification most commonly occurs during snowmelt or precipitation that results in rapid increases in the discharge rates of surface waters. Typically the decrease in ANC and pH of streams and lakes is accompanied by increases in Al^{n+} concentrations (2). These decreases in ANC and pH and increases in Al^{n+} concentrations have marked effects on water chemistry and freshwater biota.

RELATIONSHIP TO CHRONIC ACIDIFICATION

The importance of surface water acidification has been clearly recognized as an important environmental issue that has often been linked with atmospheric deposition of acidic compounds, such as sulfuric (H_2SO_4) and nitric acid (HNO_3). It has been suggested that acidic deposition has increased the extent and severity of episodic acidification. Both chronic and episodic acidification can be due to both natural (e.g., organic acids) and anthropogenic processes (e.g., strong mineral acids that are associated with SO_4^{2-} and NO_3^-). Often, acidification episodes are most marked in those surface waters near circumneutral pH (3). For surface waters at chronically low pH levels, episodic acidification is relatively less important and in some cases may not be evident (4).

CAUSES OF ACIDIFICATION

Hydrologic Flow Paths

The chemical characteristics and the relative magnitude of an acidic episode are linked to hydrologic flow paths. Hydrologic flow paths vary with landscape features and are also influenced by antecedent moisture conditions. Flow paths affect both the chemical attributes and the reaction time of the water as it drains from terrestrial to aquatic environments. Major hydrologic events such as snowmelt or large rainstorms can cause marked changes in discharge rates and rapid changes in surface water chemistry. Periods of high discharge are accompanied by dynamic changes in the size of water pools (e.g., riparian areas, soil water), flow paths and the relative contributions of different water sources to surface water. A considerable focus has been placed on evaluating the importance of snowmelt episodes in affecting surface water chemistry, especially in those areas of high amounts of acidic deposition and extended periods of snow accumulation.

During snowmelt episodes, the most marked changes in discharge rates often occur during "rain on snow." Such events can result in rapid movement of solutes to surface waters. In addition, flow paths (e.g., water flux via macropores) may develop during high soil moisture conditions that result in little contact of storm water in soil neutralizing processes and thus exacerbate acid pulses. In contrast, during dry conditions, H^+ and mobile anions (SO_4^{2-} and NO_3^-) can accumulate in soils and these ions will be flushed from the soil only during large rainstorms.

Chemical Relationships

Chemical factors affecting episodic acidification vary spatially and temporally among watersheds due to differences in geology, soils, climate, landscape features, and anthropogenic influences.

Dilution of Base Cations. In circumneutral streams that have relatively high concentrations of base cations (especially Ca^{2+}), the dilution of these base cations (as much as 100 to 280 $\mu\text{mol}_e \text{ L}^{-1}$) due to rapid increases in water discharge can contribute to rapid declines in ANC.

Sulfate. Pulses of SO_4^{2-} play a major role in episodic acidification in Europe, Canada, and some limited regions in the United States. The importance of SO_4^{2-} in contributing to acidic pulses has been most clearly shown for Europe, especially in those regions that have had much higher atmospheric inputs of sulfur than generally found in North America.

Nitrate. Fluxes of NO_3^- in contributing to acidification are most important in the northeast United States including the Adirondack and Catskill Mountains of New York State. Pulses of NO_3^- are mostly attributed to snowmelt episodes, but marked losses of NO_3^- can also occur during large precipitations that follow extended periods of low water availability. Some of the NO_3^- is derived directly from the snowpack, but most of NO_3^- comes from the soil which has accumulated NO_3^- from the microbially mediated processes of ammonification (N mineralization) and nitrification.

DOC. The contribution of organic acid (DOC) is important in many regions, and its relative role is most evident at those sites where wetlands are notable. Organic acids have been found especially important under saturated conditions in those catchments that have large proportions of peatlands (5).

Sea Salt Effect. The salt effect is an important contributor to chronic acidification in coastal areas of the northeast United States and western Europe. The deposition of neutral sea salts in coastal areas can cause pH and ANC depression (Fig. 1).

MODELING OF EPISODIC ACIDIFICATION

Various types of models have been used in the study of episodic acidification, including simulation and empirical approaches. A large number of physically and chemically

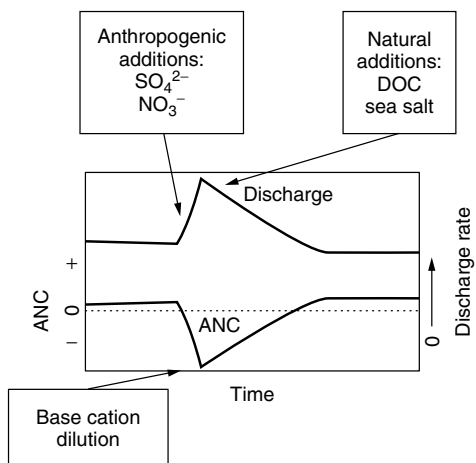


Figure 1. Schematic diagram of factors affecting episodic acidification and changes over time in acid neutralizing capacity (ANC) and discharge rate.

based simulation models have been used to predict acidification (6). Empirical models include four types: solute concentration versus stream discharge, mixing models (such as EMMA, end-member mixing analyses),

time series relationships, and multiple linear regression models. The regression model approach has been particularly useful for regional predictions of which surface water sites may be most susceptible to acidic episodes (7).

REGIONAL DIFFERENCES

Episodic acidification has been well documented for the United States, Canada, and Europe. The causes of episodic acidification vary among geographical regions. Within the United States, most of the surface waters showing episodic acidification are in the Northeast and all regions document the importance of base cation dilution. Organic acid (DOC) contributions are most important in the Adirondack Mountains. The importance of NO_3^- is most evident in the Adirondack and Catskill Mountains. In Pennsylvania, changes in SO_4^{2-} concentrations are most important in affecting both episodic and chronic acidification (3). Similarly for Europe, SO_4^{2-} appears to be most often associated with acidic episodes, although in some sites, NO_3^- , DOC, and sea salt may also be important (1). Research in southeastern Canada has suggested the importance of base cation dilution in circumneutral waters compared to low ANC waters where SO_4^{2-} is the primary controller of acidic episodes.

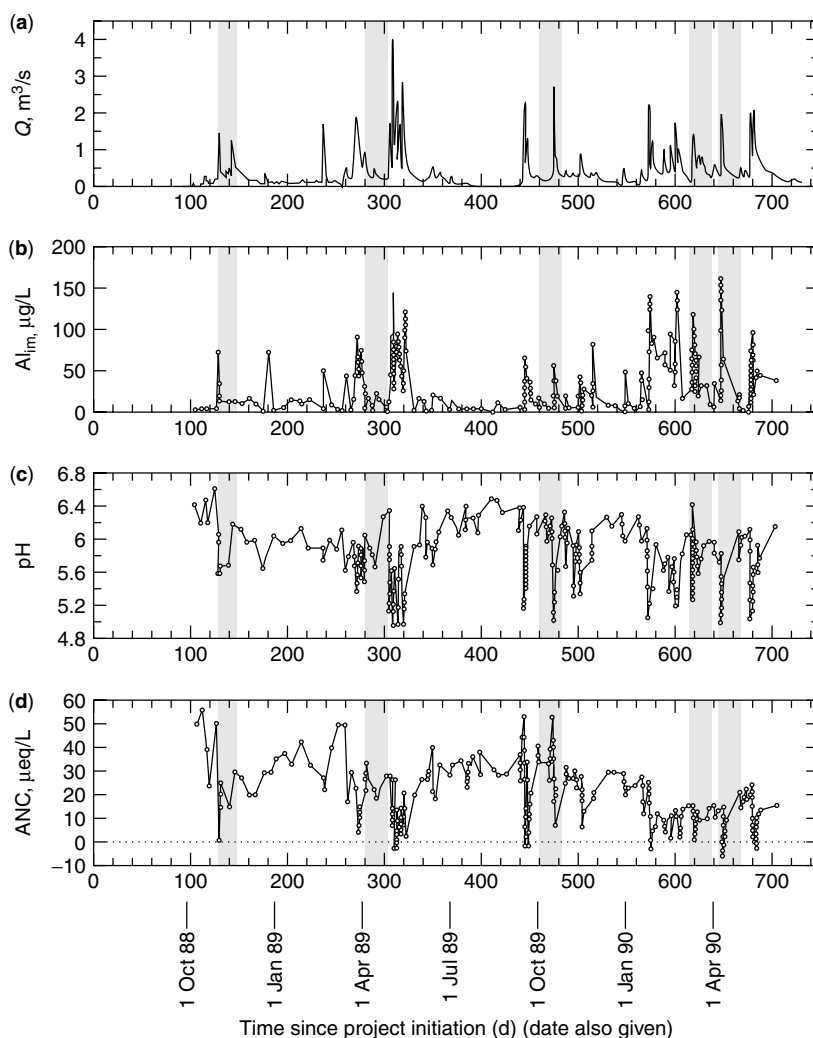


Figure 2. Effects of seasonal changes in discharge in affecting Al_{im} , pH and ANC. High discharge and low ANC coincide with spring snowmelt and fall storms. (a) Mean daily discharge (Q); (b) Al_{im} (inorganic monomeric aluminum); (c) pH; and (d) acid neutralizing capacity (ANC) for Biscuit Brook in Catskill Mountains, New York, U.S. Shaded areas are times when trout bioassays were conducted (3).

BIOLOGICAL IMPORTANCE

Episodic acidification associated with high concentrations of inorganic monomeric aluminum (Al_{im}), it has been shown, is deleterious to aquatic biota (8). In many regions, there is a general relationship between low pH, low ANC, and high concentrations of Al_{im} , but those surface waters that have lowest pH values do not necessarily have the highest Al_{im} values. The concentration of Al_{im} , it has been shown, is a good predictor of fish mortality. There are, however, major differences in the sensitivity of fish species to acidification, including acid pulses. Some acidic events may coincide with particularly sensitive periods of aquatic biota life cycles (e.g., spawning), so these events may have marked impacts on freshwater ecosystems. Other biotic impacts can include changes in the structure of aquatic communities and losses in biotic diversity (Fig. 2).

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ACIDIFICATION OF FRESHWATER RESOURCES

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GENERAL ASPECTS

All freshwater resources, groundwaters, lakes, and rivers, can be acidified. Inland waters receive acid inputs via

the atmospheric path and by direct inflows from surface runoff and from groundwaters. Sources of acids are smoke from burning of fossil combustibles and from volcanoes, acidic wastewater from industrial plants, geogenic acids from weathering of sulfidic deposits at their natural sites, or from oxidation of sulfides with/after mining ores, hard coal, and lignite (1). Accordingly, the acid inputs into ground- and surface waters are either distributed by acid rain over long ranges and large areas, or they are restricted to local impacts by draining of mines, dumps, or sulfidic natural deposits. Both pathways are found in active volcanoes regionally releasing volatile acids into the atmosphere and being direct, local sources of highly mineralized acidic brines. Last but not least, freshwaters can be directly acidified by inflows of industrial wastewater. The effect of acidic inputs is larger in soft water because of the low content of ions and low buffering capacity.

The primary state of acidic waters is changed by secondary processes in soils and rock while seeping through the ground. Thereby, the ionic composition is modified by partial neutralization of the initial acidity and by increased mineralization at low pH. Due to the processes connected with ground passage, the waters usually contain additional dissolved substances, aluminum, iron, and other elements.

The atmospheric pathway distributes anthropogenic acids stemming from gaseous SO_2 and NO_x over large areas of land. During soil passage, this rain acidity can be neutralized in carbonate-rich areas, so that, here, ground- and surface waters are not affected. However, regions that are poor in carbonates (granitic bedrock, gneiss, sand, and sandstone) are sensitive. In a two-step process, the soils first become acidic after complete loss of carbonatic minerals; then, further acidic waters seeping through the soils dissolve further minerals, especially Al silicates.

Acids of geogenic origin can be set free by natural weathering or by weathering processes induced by mining. The first geogenic source of acidity is sulfur (metal sulfides: pyrite or other sulfidic ores) that can be oxidized after contact with air. Seeping water produces sulfuric acid and Fe (II), that later can be oxidized to Fe (III), thereby causing further acidity. The processes of sulfur and iron oxidation are accelerated by sulfur and iron oxidizing bacteria. This type of geogenic acidification is common in many mining areas where we find acidic open pit lakes, acidic streams, or acid mine drainages and tailings.

Another geogenic source of acidity is volcanic activity. Volatile acids are thermally set free as gaseous SO_2 , HCl, and HF from the respective minerals. By mixing with steam of meteoric waters, strong acids are formed that dissolve volcanic rock. The resulting outflowing waters are extremely acidic and highly mineralized brines that often emerge in geothermal hot springs at the flanks and top of the volcanoes or collect in crater lakes. Because of the respective dominating buffering systems in the different acidic waters, the pH values are kept within typical ranges (2–5).

Ranges of Proton Concentration, Acidity, and Dissolved Solids

Generally, the content of dissolved minerals that are hydrolyzed during the passage of water through rock and soil increases with the acidity and proton concentration ($-\text{pH}$) of the resulting acidic waters. The total dissolved solids (TDS) increase, over the range from pH 4 to pH 0, from 200 mg/kg to 100 g/kg (Fig. 1).

The relationship between acidity and pH is shown in Fig. 2 for volcanic waters, for pit lakes, and for rain-acidified soft-water lakes. The log-log line is valid for strong acids completely dissociated into protons and anions [$\log_{10}(\text{acidity in eq/L}) = (-\text{pH})$]. The real values of the actual proton concentrations, however, are lower in the acidic volcanic waters where only 30% of the acidity is apparent from the pH. In mining waters of 10 to 30 meq/L acidity, only 12 to 30% of the protons that potentially can be set free are reflected by the pH values. This means that the acidification effect after mixing and dilution with fresh waters is three to eight times stronger than presumed from the actual pH values.

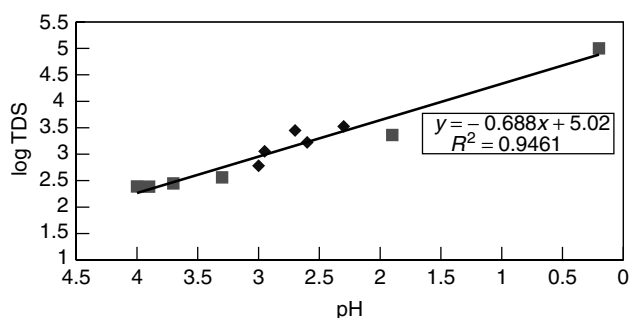


Figure 1. Relationship between the pH of acidic waters of different types and the content of total dissolved solids (TDS in ppm): diamonds: original values for pit lakes; squares: values for volcanic waters from Reference 6.

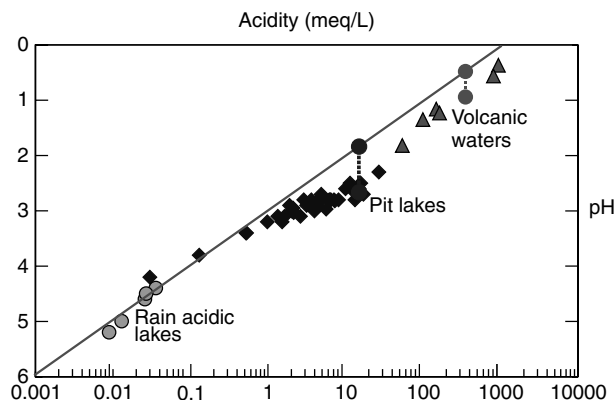


Figure 2. Acidities (measured as base neutralization capacity ($K_{B4.3}$) or calculated from an ion balance) and pH values for acidic volcanic waters (triangles: values from Reference 6), for acidic pit lakes (diamonds: original values and from Reference 7), and for rain-acidified softwater lakes (circles: from Reference 8). Vertical stippled lines show the deviation between the actual proton concentration (pH) and the acidity (or base neutralization capacity) of the waters: in pit lakes, 12 to 30%, and in volcanic lakes, 30% of the acidity is apparent as free H^+ ions.

ATMOSPHERIC ACIDIFICATION: ACID RAIN AND SENSITIVE WATERS

Acidification of Soil and Water and Chemistry of Acidified Soft Waters

Prior to anthropogenic emissions, the only atmospheric acid was carbon dioxide, resulting in weakly acidic rain of pH 5.6. This natural rainwater dissolves only carbonates from soils and weathering bedrock. Therefore, natural freshwaters contain mainly carbonates, HCO_3^- , and CO_3^{2-} , and most fresh inland waters in temperate climatic regions show nearly identical chemical compositions in which the carbonate buffering system dominates at pH 7. Carbonate-poor geological regions have “soft” waters of low mineral content, and carbonate-rich areas show higher mineralized “hard” waters. The relative (!) ionic composition of all natural freshwaters is similar, resulting in a “standard water composition” (2,9).

Only those geological regions that are poor in carbonates are sensitive to acid rain. The critical load is reached when the base saturation (percentage of Ca^{2+} , Mg^{2+} , K^+) decreases below 20% of the total cation exchange capacity (10) or below a limit of acid neutralization capacity (ANC) of 0 to $25 \mu\text{eq L}^{-1}$ (11). Estimates of the critical loads (11) showed a range between 300 and 800 mg m^{-2} of annual sulfur depositions in geologically sensitive areas (12–15).

Acid rain changes the chemical composition of seeping runoff, ground- and surface waters (8,11). Beyond the critical limits, the interaction between soil and rainwater leads to accumulation of acidity (N and S content). The carbonate and hydrogen carbonate anions in soil and surface waters are replaced by sulfate, and Ca^{2+} and Mg^{2+} cations by H^+ and Al^{3+} (16). Rain-acidified inland waters, after a large-scale process of titration, are carbonate-free, enriched in sulfate and aluminum, and acidic within a range between pH 4.5 and 5.5. In regional surveys of sensitive areas, the frequencies of pH of the respective lakes show bimodal distributions (3,17–20). Two buffering systems are involved. The nonacidified lakes are circumneutral and buffered by the carbon–acid system ($\text{CO}_2\text{--H}_2\text{CO}_3\text{--HCO}_3^-\text{--CO}_3^{2-}$). The rain-acidified soft waters are moderately acidic (pH 4.5–5.5) and are buffered by the aluminum system ($\text{Al}^{3+}\text{--Al(OH)}^{2+}\text{--Al(OH)}_2^+\text{--Al(OH)}_3\text{--Al(OH)}_4^-$) (2,11,21).

Regional Distribution of Rain-Acidified Waters

After the emissions of acid smoke, transport times of 15 to 21 hours were observed from the U.S. Midwest states, where the emissions occurred, to the Northeast states in New England and Canada, where the acid rain was precipitating. Accordingly, models use 15-h and 21-h back trajectories to refer the observed acid rain to the emission sources (22). Similar transport times and distances are known from England to southern Scandinavia. However, longer transports are also known that enable intercontinental atmospheric transport (23). The present global problem areas at risk of surface water acidification (14) are in mid- and northeast North America; in western, central, and northern Europe (24); in the eastern parts of China (25); and in different tropical

countries (26–28). The long-term development and large-scale geographic distribution in comparison with the development in North America were described in a 9-year report of 11 EU countries (29). In Norway, 27% of the lakes were acidified, and 9% in both Sweden and Finland, corresponding to 6000 and 3000 lakes, respectively (30). Of a total number of 85,000 lakes (20%) in Sweden, 17,000 were assigned as affected until 1992 (31).

Biological Effects of Acidification on Physiological and Ecosystem Levels

The biological effects of rain acidification on individual groups of organisms and on the ecosystems were extensively studied (3,4,15,17–20,22,29,32–52). As acidity increases and pH decreases, the ecosystems lose major groups of organisms; fish, mollusks, cyanobacteria, and crustaceans. There is a shift to fewer species in all groups and to lower biodiversity. The simplified food web has some new dominating components, such as Dinophyceae, water bugs, sphagnum moss, or mats of filamentous green algae. A combination of stressors was identified: acidity, toxic species of aluminum, more intensive and deeper reaching UV irradiation, and loss of fish-food, such as macroinvertebrate species. In several thousand acidified lakes in southern Norway, the macroinvertebrates disappeared at a pH below 4.8. Many fish species disappeared at a pH below 5.7 in the acidic lakes of the La Cloche Mountains (NY), and no species survived at <pH 4.3 (17,20). About 50% of the populations of brown trout in southern Norway were lost in the respective area till 1975 (43,44). The losses increased with altitude in headwater lakes >1000 m above sea level to more than 80% (4,35). A survey showed till 1990, a total of 9,630 fish populations lost, and 5,405 populations were severely affected. The damaged area covered 25% of Norway (40). In a survey of benthic macroinvertebrates in neutral and acidified streams and in acidic lignite mining lakes in Germany, a maximum of about 50 species was found at pH 8, and the number of species decreased to zero at pH 2 (53,54).

Success of Countermeasures and Long-Term Developments

Countermeasures against acid rain and acidification of soils, lakes, and rivers started (1) by controlling the emissions of acidic smoke and (2) by direct measures applied to acidic waters, mainly liming. The emissions were reduced in North America and in Europe during the last two decades by technical improvements, as documented by monitoring networks (55) and by long-term observations at single sites (56,57). In Europe, the former Eastern countries followed after a one decade delay; the area of former East Germany could be identified by the atmospheric SO₂ content till 1992 (58). In the world's largest liming program in Sweden, 200,000 tons of lime were spread every year for 20 years. Of 6000 lakes treated, about half of the acidified area was restored till the early 1990s (31,59–61). The state of "Acidification of surface waters in Sweden—effects and counteracting measures" in 1993 is described in a series of 12 papers (AMBIO 1993, issue 12/5).

INDUSTRIAL ACIDIFICATION: LAKE ORTA (ITALY)

The input of industrial wastewater into fresh waters caused acidification and contamination. Between 1926 and 1982, prealpine Lake Orta (143 m deep, 1.3 km³ volume) was heavily polluted by industrial wastewater containing ammonium sulfate and heavy metals Cu, Cr, Ni, and Zn. The development of the pollution and acidification and of the recovery are well documented (62). The annual input of copper reached 70 tons between 1950 and 1954, resulting in mean lake concentrations of more than 0.1 mg Cu L⁻¹. The ammonium input was 3,000 tons NH₄⁺-N per year during the late 1960s. During the 1960s and 1970s, the lake water became acidic (pH 3.9 to 4.5) by oxidation of the ammonium. Fish and populations of animals disappeared nearly completely. After stopping the input of industrial wastewaters, the lake was treated from 1989–1990 by lime suspension. The lake was completely neutralized (pH 6.7–6.9) by using 18,000 tons of powdered limestone (63). To date, this was the world's biggest liming campaign applied to a single lake.

GEOGENIC ACIDIFICATION

Lakes and rivers can be acidified geogenically by volcanic acids and by natural acid tailings originating from ore and coal deposits, from surface-mining dumps, or from drainage of old underground pits. Direct acid input from mine tailings into lakes and rivers results mostly from sulfide minerals such as pyrite and related ores oxidized to sulfuric acid and iron hydroxides.

Natural Geogenic Acidification: Acidic Crater Lakes

In volcanic areas, natural pH values of acid crater lakes range from pH 0 to 5.5 (for volcanoes and their state of activities, see References: 64,65). Crater lakes of many active volcanoes contain highly acidic waters that originate from volatile mineral acids. Sulfuric acid, hydrochloric acid, and hydrofluoric acid, or their volatile forms, emerge from the hot active centers causing subsequent leaching and weathering of rock. These acid waters often collect in crater lakes with given constraints in their physical and chemical properties (66,67). For recent publications on crater lakes, see Reference 67. Several crater lakes and volcanic acid brines were chemically characterized (68–72) and showed a very broad spectrum of dissolved elements, many heavy metals, and high temperatures.

Natural Acidic Tailings from Geological Deposits

Natural acidification can originate from natural mineral sulfides weathering in soils and sediments (73). Examples of lake acidification from ore bodies or from metalliferous black shales in Japan (74) and Canada (75–77) were described. The near-surface weathering could have been initiated by glacial processes; the results are comparable with human mining activities (78). Furthermore, post-glacial isostatic land uplift can bring about long-term weathering of near-surface ores (gossans) (77). Examples of acidic hypersaline lakes in endorheic regions are located in Australia and Chile (79–81). The former existence

of ancient lakes of this type was shown in the United States (73).

Acidic Rivers—Rio Tinto

The rivers Rio Tinto and Rio Odiel in southern Spain are natural acid drainages from the Iberian Pyrite Belt, the world's largest deposit of sulfidic ores. There are indications (82) that acid waters existed in the river basin 300,000 years ago (83). In addition to the natural acid drainages, human mining activities increased acidification for about 4,500 years and especially during the last 150 years. The Rio Tinto River is an extreme aquatic environment over its full length of 92 km, it has a mean pH of 2.2 and high concentrations of heavy metals (e.g., Fe 2.3 g/L, Zn 0.22 g/L, Cu 0.11 g/L) (82). The metal pollution is reflected in the estuarine and nearshore marine sediments that contain Fe (11.2%), Cu (0.93 g/kg), Zn (1.15 g/kg), Pb (0.73 g/kg), and Ba (0.66 g/kg) (84).

Acidic Mining Lakes

Lignite surface pits in east Germany (the former GDR) were largely closed after the unification of 1989. The voids that are left or will be filled within a few years will result in 200 pit lakes >1 ha. In Lusatia, the eastern lignite mining district of the former GDR, 63% of the existing pit lakes are acidic, have pH values of 2.3 to 3.5, and show high sulfate concentrations up to 2.5 g/L (85). In the United States, surface coal mining left hundreds of pit voids and lakes in the Appalachians and the Midwest during the first half of the last century (86). In addition, numerous new lakes will originate from 86 major ore mining plants (86). Thirty new pit lakes are expected in Nevada within the next 20 years. Existing acidic pit lakes are Berkeley Pit in Montana, Liberty Pit in Nevada, and Spencerville Pit in California (86). Presently, 19 metal mines are operating in Canada, 74 in Australia, 37 in Chile, 75 in Kazakhstan, and additional ones in other countries (Brazil, Peru, Mexico, Indonesia, Philippines, and in African and Asian countries) (86). Many of the voids left will become future lakes that often will be acidic and contaminated by toxic metals.

The present state of the literature does not permit one to work out satisfactorily the common chemical characteristics and a typology of the mining lakes of the different geological regions and climatic zones. A basic classification was found in regional surveys of lignite pit lakes, where multimodal frequency distributions of pH values similar to those of rain-acidified lakes were observed. The three modal groups of pit lakes are based on three buffering systems: bi-carbonate (pH 6 to 8), $\text{Al}(\text{OH})_x$ (pH 4.5 to 5.5), and $\text{Fe}(\text{OH})_x$ [pH 2 to 4] (87), and citations in (86)].

FRESH WATERS AFFECTED BY ACIDIC INFLOWS

Nonacidified waters, rivers, lakes, and groundwater are affected by mixing with acidic inflows such as acidic rivers from volcanic sources' acid drainages and tailings from natural deposits and from surface and underground mines, or acidic sludges that are accidentally set free by spills (89).

In acidic drainages, heavy metals are often found as cocontaminants. Although the compounds are diluted, neutralized, and minerals are precipitated, acids and metals damage the freshwater biota (88). Unexpectedly diverse communities of extremophiles can develop in permanently acidic and metal-rich waters (82,83).

Acidification of Groundwaters by Acid Rain and Mining

The importance of the acidification of groundwater tables is related to the spatial scale of the affected aquifers. The effect of acid rain on springs and brooks in Germany was observed by a monitoring network whose areal distribution was screened by a 3-km grid (90). The results showed acidification of all near-surface groundwaters in regions sensitive because of soils poor in carbonates. The scale of lignite mining in the east German district of Lusatia also generally affects regional groundwater (91).

Acidification of Rivers and Lakes by Volcanic Water

Active volcanoes set free volatile acids (SO_2 , HCl, HF) and form acidic brines. Outflows of the resulting acidic crater lakes and volcanic flank springs lead to highly acidic headwaters in catchment areas. Freshwater streams that receive the acidic and toxic volcanic inputs of the headwaters are heavily affected (65). Two examples from Java show the consequences of this sort of pollution. The crater lake of the volcano Kawah Ijen, East Java, contains $32 \times 10^6 \text{ m}^3$ of hot and acidic brine (pH <0.4, TDS > 100 g/kg, SO_4^{2-} 70 g/kg, Cl^- 21 g/kg, F^- 1.5 g/kg). The brine seeps through the crater rim and mixes with the Banyupahit River, where the acidic inflow is only incompletely neutralized by dilution and results in the lack of biota. About 30 km downstream, rice fields are irrigated by the river, discharging 4,000 L/s; the daily load is 150 t SO_4 , 2.8 t F, 50 t Cl, 10 t Al, 35 kg Ti, and 4 kg Cu (92). The age of the crater lake and its environmental impacts are estimated at more than 200 years old. Furthermore, the Patuha volcano in West Java has an acidic crater lake, Kawah Putih, and springs of acid brines of pH < 1 containing Al, As, B, Cl, Fe, Mn, and SO_4 . The acid streamlets drain into the Citarum River that is contaminated by these potentially toxic elements from the crater lake and, too, is used for irrigation (93). In Argentina at Copahue Volcano, there is a network of acidic waters starting from an acid crater lake, Lago Copahue, and a 13 km-long, extremely acidic river, Upper Rio Agrio (pH 0.6 to 1.6). The river discharges into the glacial Lake Caviahue, where the highly acidic water is diluted to pH 2.5 (68,94). The outflow of the lake, Lower Rio Agrio, is further diluted in its course, but, presumably, damages downstream freshwater stretches by its acidic inputs.

Acidification of Surface Waters by Mine Drainage

In the United States, Kleinmann (1989) reported that 19,300 km of streams are affected by acid mine drainage (AMD). In Europe, a typical case is the Avoca River in southeastern Ireland which is affected by continuous AMD from an abandoned copper and sulfur mine area (88). The AMD of pH 2.7 discharges annually about 300 tons of metals into the river (108 t Zn, 276 t Fe, 6 t Cu, 0.3 t Cd).

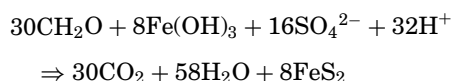
In the contaminated river, whose pH is 5.8, macrophytes and fish are eliminated, and macroinvertebrates survive only for short periods outside the mixing zone. The damage to the indigenous biota is due to a combination of metal toxicity, sedimentation, acidity, and salinization (88).

Remediation of AMD and Affected Waters

The acidification of lakes and rivers by acid drainage can be prevented by preclusive measures and direct treatment of the polluting waters as (1) restrictions on running mining activities; (2) "active treatment" of the AMD by addition of alkaline chemicals, often in on-site reactor systems, neutralizing the waters; (3) "passive treatment" using natural geochemical and biological reactions to reduce acidity, pH, and ion concentrations; and (4) insitu treatment of acidic lakes.

Gray (95) developed a scheme to describe (1) the damage of AMD on affected river systems for the site of the Avoca tin mine in Ireland and (2) a management system to avoid damage by a combined application of prevention and remediation by passive treatment methods (88). Gazea et al. (96) published a comprehensive review on treating AMD by using passive systems, such as natural geochemical and biological processes to improve water quality with minimal requirements. The remediation of small-scale acid drainages was reached by sequential flow (1) through anoxic limestone drains for pretreating acidic waters by providing alkalinity, (2) through aerobic wetlands to remove iron through oxidation, and (3) through a series of anoxic cells in compost wetlands to raise the pH and to reduce concentrations of other metals and sulfate (96–100).

In extremely acidic pit lakes, the acidity and content of dissolved substances is one to three orders of magnitude higher than in rain-acidified waters. So, liming is far more expensive, and biological treatment becomes a promising alternative for neutralizing the acidity. The microbial processes (101) used for remediating lakes by insitu methods are summarized by Klapper et al. (97). Sulfate-reducing bacteria can use sulfate as an electron acceptor to reduce both iron and sulfur to form pyrite (102):



Analogous to the process in anoxic cells of passive AMD treatments, the in situ reversion of pyrite oxidation to neutralize the acid water in mining lakes can be promoted by adding organic substances that lead to anoxic zones in and above the bottom sediments or in the hypolimnic waters. A pilot project using in situ remediation techniques on whole lakes was actually run (103,104).

DATA SOURCES

Data and actual reports on inland waters and relevant environmental issues are accessible for different regions and on a global scale from Internet sources (Table 1).

Table 1. Internet Access to Data, Reports, and Publications on Inland Waters of the World

Canada Centre of Inland Waters (CCIW)	http://www.cciw.ca/gems
European Environment Agency (EEA)	http://themes.eea.eu.int
European Union (EU)	http://europa.eu.int
Food and Agricultural Organisation of the United Nations (FAO)	http://www.apps.fao.org
Global Runoff Data Centre	http://www.bafg.de/grdc.htm
Global Water Partnership	http://www.gwpforum.org
Organisation for Economic Cooperation and Development (OECD)	http://www.oecd.org
World Hydrological Cycle Observing System	http://www.wmo.ch
U.S. Environmental Protection Agency (EPA)	http://www.epa.gov

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GEOCHEMISTRY OF ACID MINE DRAINAGE

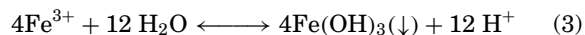
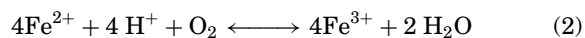
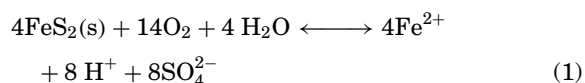
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Acid mine drainage (AMD) results when metal pyrites come in contact with water and/or air, to form dilute sulfuric acid. For instance, iron pyrite (FeS_2) is the major iron-sulfur impurity found in mined earth, particularly in the eastern United States where coal mining is prevalent. Typically, AMD waters contain elevated concentrations of SO_4 , Fe, Mn, Al, and other metal ions. As an example, representative AMD water quality parameters from the Roaring Creek-Grassy Run Watershed, located in Elkins, West Virginia, include pH from 2.4 to 3.3, mineral acidities from 2.4 to 980 mg/L as CaCO_3 , dissolved iron between 35 and 260 mg/L, and sulfate concentrations from 190 to 740 mg/L.

In contrast, copper and arsenic sulfide compounds associated with “hard rock” mining operations are common in the western United States; such compounds are generally much less prevalent in the east. In such instances, sulfide containing minerals such as pyrrhotite (FeS), arsenopyrite (FeAsS), and chalcopyrite (CuFeS_2) can produce acidic drainage when oxidized.

Stoichiometrically, the interaction of iron pyrite with water is described by the following reactions:



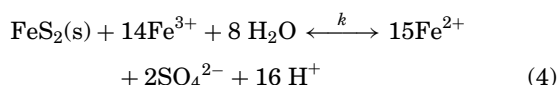
In this process, iron pyrite is oxidized to produce ferrous iron (Fe^{2+}), protons and sulfate ions. Ferrous iron is subsequently oxidized to ferric iron (Fe^{3+}) and precipitates from solution as $\text{Fe}(\text{OH})_3(\downarrow)$, the sulfate ions can combine with protons to produce sulfuric acid. The iron solids, which form the characteristic red-yellow/yellow-brown coating on stream beds, can consist of a variety of ferric oxides, hydroxides, or oxyhydroxysulfates, including ferrihydrite [$\text{Fe}_3(\text{OH})_4$, $\text{Fe}_5\text{HO}_8\text{H}_2\text{O}$, or $\text{Fe}_5\text{O}_3(\text{OH})_9$] and schwertmannite [$\text{Fe}_8\text{O}_8(\text{OH})_6\text{SO}_4$] (1–5). Further, ferrihydrite and schwertmannite are metastable and may eventually dehydrate and recrystallize to form hematite (Fe_2O_3).

Many factors determine the rate of AMD generation, including the presence and activity of bacteria, pH, temperature, and pyrite surface characteristics. The interactions of these factors are complex and readers are

referred to the following for additional information on AMD geochemistry: Temple and Koehler (6), Singer and Stumm (7), Kleinmann et al. (8), Nordstrom (9), Hornberger et al. (10), Alpers et al. (11), and Evangelou (12). Further, approaches to managing/remediating various sources of AMD are largely functions of those parameters that govern the rate of AMD generation. Representative sources of information on the prevention and treatment of AMD include Sobek et al. (13), Faulkner and Skousen (14), Skousen and Ziemkiewicz (15), Caruccio et al. (16), and Meek (17).

Bacteria and pH

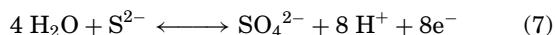
In many cases, the presence and activity of bacteria control the rate at which AMD forms. For example, in addition to the series of reactions presented previously, iron pyrite can interact with ferric iron according to the following relationship, in which additional acid and dissolved iron ions form:



The rate law expression for the oxygenation of ferrous iron is given as

$$-\frac{d[\text{Fe}^{2+}]}{dt} = k[\text{Fe}^{2+}]p_{\text{O}_2} \quad (5)$$

where p_{O_2} is the partial pressure of oxygen and k is the reaction rate constant. The reaction rate constant, k , is $\approx 10^{-25}$ L/atm·min at 25 °C at pH < 5.5. Thus, iron oxidation is predicted to be very slow at pH < 5.5. However, it is known through field and laboratory studies that pyrite is rapidly oxidized to Fe^{3+} at pH values of 2 to 3. In this case, the microorganisms *Thiobacillus thiooxidans*, *Thiobacillus ferrooxidans*, and *Ferrobacillus ferrooxidans* catalyze the oxygenation of ferrous iron (18). In particular, *Thiobacillus ferrooxidans* catalyzes or mediates the following oxidation–reduction reactions of Fe^{2+} :



Similarly, *Ferrobacillus ferrooxidans* mediates the Fe^{2+} – Fe^{3+} reaction presented in Eq. 6.

To develop an understanding of the effects of pH on AMD formation, it is necessary to consider the solubility of Fe^{3+} hydroxide species in water. At equilibrium, the relationship between the $\text{Fe}(\text{OH})_3$ solid, soluble Fe^{3+} species [presented as $-\log(\text{soluble iron species})$] and pH is given in Fig. 1.

Kleinmann et al. (8) and Nordstrom (9) presented a three-step process of chemical and biologically mediated reactions through which AMD is generated. Note that this sequence is based on processes in the absence of appreciable amounts of alkaline material, in unsaturated systems, where there is an adequate oxygen supply. Thus, applying this rationale to systems that have substantially different characteristics must be done with caution. Stage

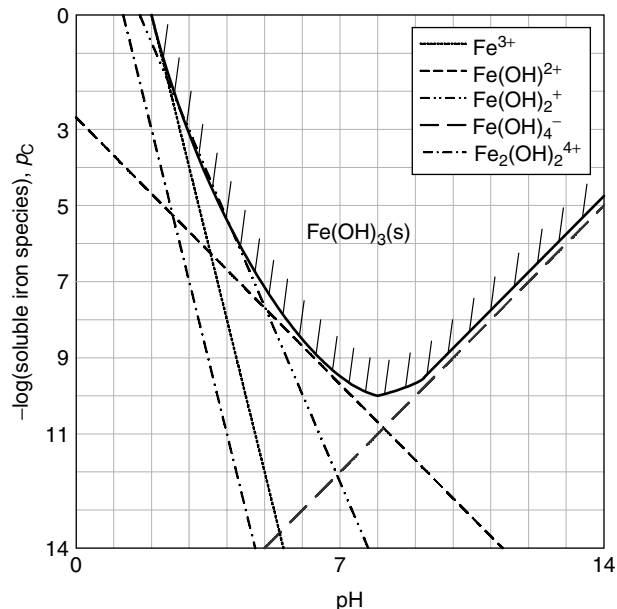


Figure 1. Log concentration versus pH for soluble Fe^{3+} species in the presence of $\text{Fe}(\text{OH})_3(\text{s})$.

I occurs at circumneutral pHs, where pyrite is oxidized chemically, as presented in Eqs. 1 and 2. Under such conditions, the Fe^{3+} concentration is limited by the low solubility of $\text{Fe}(\text{OH})_3$.

However, some limited oxidation of pyrite by bacteria may occur on the surface of pyrite grains. In Stage II, which occurs at pHs from 3.0 to ~4.5, Fe^{2+} chemical oxidation is slow; consequently, oxidation is mainly bacterially mediated. In Stage III, the relationship presented previously in Eq. 4 is principally responsible for acid production, which occurs much more rapidly than that in Stages I or II. Typically, Stage III occurs at pH < 3 as Fe^{3+} becomes increasingly soluble, and the rate of pyrite oxidation can increase rapidly, as presented earlier. However, for pH < 1.5 to 2, the effectiveness of *Thiobacillus ferrooxidans* as a catalyst for the oxidation of Fe^{2+} decreases (19,20).

One exception to the low pH conditions characteristic of most mine waters includes the case where carbonaceous minerals such as calcite and dolomite impart alkalinity to the water that equals or exceeds the acidity. Dissolution of carbonaceous minerals produces alkalinity, which neutralizes acidity and promotes the removal of Fe, Al, and other metal ions from solution. The result is drainage that is either neutral or net alkaline, though such waters can still have elevated concentrations of SO_4 , Fe, Mn, and other AMD-associated metal ions and may require further management.

PROPERTIES OF PYRITE

The rate of acid production depends on the surface area of pyrite exposed to a water solution or air; pyrites with higher surface areas produce more acid than pyrites with low surface areas (21). Similarly, rocks with a high percentage of pyrite produce acidity more rapidly

than rocks with a low percentage of pyrite. Further, conclusions regarding the influence of surface defects, such as pitting, on pyrite oxidation rates have been studied by McKibben and Barnes (21). Other investigators have studied the crystal structure of various pyrite samples to explain differences observed in the rates of oxidation among multiple samples. For instance, Kitakaze et al. (22) observed vacant positions in the crystal lattice of pyrite and related variations in such physical properties to differences in oxidation rates. However, further studies are needed to elucidate the specific effect(s) that lattice structure, atomic spacing, and so on, have on pyrite oxidation rates.

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THE ARAL SEA DISASTER: ENVIRONMENT ISSUES AND NATIONALIST TENSIONS

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AN UNPRECEDENTED DISASTER UNFOLDING

The village of Muynak was located on an island in the Amu Daria delta in 1956. In 1962, the island had already become a peninsula; in 1970, the sea was 10 km away, then 40 km away in 1980, and 75 km in 1998. The sea bottom became a vast salted desert. In 1987, the retreating sea divided itself in two parts, a small Aral in the north, where the Syr flows, and a large Aral, where the Amu ends its journey when it reaches the sea (Figs. 1 and 2).

Between 1960 and 1998, the level of the small Aral dropped by 13 meters, and the level of the larger by 18 m. The total sea volume was cut by 78% (5,6). The vanishing of the sea constitutes an enormous ecological disaster that radically alters the environmental balance in the region:

- Vast deposits of salts, nitrates and pesticides spread in large quantities on the industrial crops upstream were formed, deposited by water of the Amu and the Syr. The wind now blows on these desiccated deposits, lifting nearly 40 million tons of toxic sediments



Figure 1. The watersheds of the Amu and Syr Rivers (1,2).

each year, and transporting them far inland: this phenomenon poisons the soil, the vegetation, and the inhabitants. The salt fallout can be as high as 500 to 700 kg/ha, which quickly worsens the problem of salt built-up due to poorly drained and excessive irrigation.

- As the Aral Sea, that acted as a climate regulator, gradually disappears, the local climate takes on distinct continental characteristics:
 - The frost-free period was shortened to about 170 days that undermines the profitability of various crops, including cotton, which needs about 200 frost-free days. As a consequence, some farmers have turned to growing rice, a crop that consumes even more water. Summers are shorter but also hotter.
 - Strong winds are more frequent, increasing the amount of toxic sediment that is blown inland. Aral dust has been detected as far away as Belarus, about 2000 km to the West.

- Local rain levels dropped by half in the sea area, and relative humidity dropped by about 30%.
- The salinity of the sea increased from 9.9 g/L in 1960 to 30 g/L in 1994, destroying most of marine life, then to 45 g/l in the larger Aral in 1998 (5). In 1977, the fish catch had already decreased by 75%; in 1982, all commercial fishing had stopped.
- The retreating sea level made the main aquifer level drop, thus drying up several oases around the sea area (7,8).
- Intensive and improperly managed irrigation led to salinization in as much as 95% of cropland (8 million hectares only for cotton), which drives the farmers to increase the amount of fertilizers and water (9,10), thus compounding the water stress.

The Aral Sea, the fourth lake in the world by its surface in 1960, went into a steady decline (Table 1).

On a geologic timescale, changes in the Aral Sea level have been noted: the phenomenon is not, in itself, radically new. The sea is naturally sensitive to variations as it is not very deep (16 m on average) and subject to strong evaporation (about 58 km³/year, on average). But it is the speed with which the Aral Sea disappeared from 1960 on that hints of human causes, which probably were compounded by natural causes. A sharp decrease in precipitation was observed in the 1970s and 1980s; by itself, it was estimated that it would have led to a decrease in the sea level from 53.5 m in 1960 to 50.8 m in 1986. The rest can be accounted for by large withdrawals for irrigation, a demand all the stronger as rain became scarcer.

DEVELOPING THE VAST CENTRAL ASIAN PLAINS

The Russian Empire introduced cotton in Central Asia in the late nineteenth century. But it became intensively grown under the Soviet regime. In 1937, the Soviet Union, eager to increase its export revenues, became a net cotton exporter. There were social and political goals as well: through the collectivization and the mechanization that were put in place so as to expand large industrial cotton

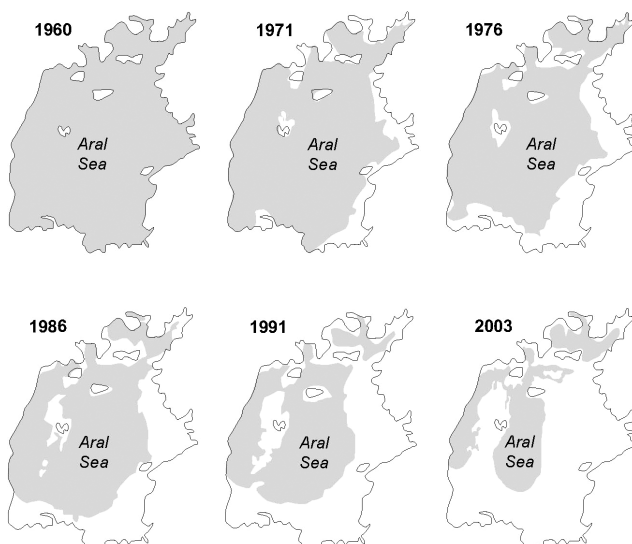


Figure 2. Evolution of the Aral Sea from 1960 to 2003 (3,4).

Table 1. State of the Aral Sea from 1950 to 1998^a

Year	River Discharge, km ³	Sea Level, m Large Aral After 1987	Surface, km ²	Volume, km ³	Salinity, g/L Large Aral After 1987	Fish Catch, t
1945	62				10	
1960	40	53.41	66,458	1090	10	43,300
1965	31	52.5	63,900	1030	10.5	31,040
1970	33	51.6	60,400	970	11.1	17,460
1975	11	49.4	57,200	840	13.7	2,940
1980	4	46.2	52,500	670	16.5	0
1985	0	42	44,200	470	23.5	0
1987	0	40.5	41,000	404	26.8	0
1989	6	38.6	36,900	330	30.1	0
1990	9	37.8	34,800	304	33.3	0
1993	14 (est.)	37.1	35,424	297	35	0
1998	21 (est.)	31.5	31,541	236	45	0

^aReference 3, pp. 182, 191; Reference 6, p. 15.

farms, traditional family farming was destroyed, and with it the remnants of resistance of Central Asian societies toward communism and a centrally planned economy.

Irrigation: The Main Culprit

Water was the key to a vastly expanding cotton production. Canals and dams were planned from 1950 on and built to take the water from the Amu and the Syr to the fields (Table 2). In 1956, the Karakum Canal was inaugurated. It was to irrigate the dry lands of Turkmenistan, when irrigation was considered the solution to humankind's development. In 1980, the highest dam in the world, the Nurek Dam (335 meters high), was commissioned in Tadjikistan.

Thanks to this infrastructural effort, irrigated land grew between 1965 and 1986 at an average rate of 2.1% annually (Table 3). During the same period, water demand

doubled. A lot of it was lost through evaporation in open canals or through seepage in uncemented aqueducts. Water drained the fertilizers and the pesticides sprayed over the fields, but smaller amounts reached the sea as more canals were built to develop more acreage. In 1950, 50 km³ of water flowed into the Aral; in 1990, this volume had dropped to about 7 km³; the 1990s seem to have been wetter and, combined with the beginning of irrigation rationalization, contributed to an increase of river discharge into the sea. But the amount of water now reaching the Aral is not even enough to stop the sea from retreating, let alone to consider restoring the former Aral, for which at least 35 km³ would be needed on a regular basis.

After the collapse of the Soviet Union, the Central Asian republics, which were not so eager to become independent, fell into a deep recession. Their main market, Russia, was itself in the throes of economic disarray. Trade was now

Table 2. Development of Irrigated Surfaces, Central Asian Republics, in Thousand Hectares^a

	1950	1965	1970	1975	1980	1985	1987
Uzbekistan	2276	2639	2696	3006	3476	3930	4109
Kirghistan	937	861	883	911	956	1009	1028
Tadjikistan	361	468	518	567	617	653	675
Turkmenistan	454	514	643	819	927	1107	1224
Kazakhstan	1393	1368	1451	1648	1961	2172	2318
Total	5421	5850	6191	6851	7937	8871	9354

^aReference 3, p. 146.

Table 3. Irrigation in the Aral Sea Basin^a

	Irrigated Surfaces (Million ha)	Withdrawals for Irrigation (km ³)	Withdrawals Per Surface Unit (m ³ /ha)	Surfaces Sown with Cotton ^b (ha)	Cotton, % of Irrigated Surfaces ^b
1965	4.8	82	17,000	2287	47.6
1980	6.3	107–126	17,000–20,000	2869	45.5
1985	7	112–133	16,000–19,000	3051	43.6
1990	7.25	109	14,600–17,000	2909	40.1
1995	7.94	100	12,594	2574	32.4

^aReference 6, p. 28.

^bFigures from 1993 for cotton production.

to be done at world market prices in convertible currencies, which was not favorable for them, and they realized they were highly dependent on one crop, cotton. Oil and gas reserves were already known at the time, but few pipelines existed to world markets, and all of them went through Russia, which levied a heavy transit fee (11). In 1992, cotton occupied 60% of the cropland in Uzbekistan. In 1998, the crop still represented a third of all Uzbek exports and the main source of hard currencies. Agriculture is 43% of the GDP in Uzbekistan, 46% in Turkmenistan and 33% in Kazakhstan, and employs 22% of the workforce in Uzbekistan. The Kazakh, Turkmen, and Uzbek governments, where most of the crops were grown, thus realized how difficult it would be to reform irrigation practices drastically as the more pressing problems were hardly environmental but the very survival of the economy.

Besides, available water is already insufficient to sustain all crops in the region. About 25% of the irrigated lands in Uzbekistan now receive only about 70% of the water needed for optimal cotton growth (12). But the worst is still to come. The cotton production levels induced the destruction of the Aral, but they are not even sustainable because of salt intrusion that is often fought back by spraying even more water. In 1994, 28% of the irrigated lands in the Aral basin, as against 23% in 1990, were plagued by salt deposits that reduced their productivity by 20 to 50%. Yields were adversely affected. Between 1981 and 1986, cotton production dropped by 21.3% in the Soviet Union, largely caused by salt buildup and poor management. To make up for this lost production, several countries opted to wash out the fields, which was temporarily effective but induced an even faster rise of the water table; irrigated surfaces were also increased, compensating the decreasing yield by larger surfaces, especially in Turkmenistan, where a rise of 31% occurred between 1990 and 1994.

Saving Water?

Irrigation is responsible for about 93% of the water consumption in the two Aral Sea basins. Solutions to the crisis have been known for long: they require a sharp increase in water yield and the reduction, if not the elimination, of the most consuming crops, such as rice, and their replacement with less demanding cereals, such as sorghum or millet (savings expected: 3 km³). Unprofitable crops could be phased out (a water saving potential of about 15 km³). The improvement of agricultural techniques in the most profitable cotton fields, notably by introducing more effective irrigation techniques such as sprinklers, could add an additional 8 km³. The repair, lining, and tightening of canals could help save about 10 km³.

But these solutions are all costly, whether financially or socially. Besides, they imply cooperation among five governments to implement water sharing and management schemes.

AFTER INDEPENDENCE, THE LONG ROAD TO AN AGREEMENT

When the Central Asian republics were federated states under the authority of the Soviet Union, arbitration

and infrastructural development decisions were made in Moscow. The development of the Aral Sea basins through intensive irrigation was not questioned at the time. When independence took place, the resource sharing designed during Soviet times appeared all the more unjust as downstream users, which could benefit from irrigation, pressured the two upstream countries, Kirghizstan and Tadjikistan, not to interfere with water resources stored in reservoirs located in their territories; these very downstream users are also well endowed with hydrocarbon resources. Suddenly becoming independent in 1991 due to the collapse of the Soviet Union, the five republics were forced to face bitter disputes and look for cooperative schemes.

There were tensions, even before 1991. In 1989, clashes took place between Kirghiz and Tadjik farmers over water allotments. In the Osh area in Kirghizstan, another clash between Uzbeks and Kirghiz caused 300 deaths in 1990. Along more and more closed borders, patrols are organized to prevent border crossings and irrigation canal sabotage.

Bitter disputes have emerged between Turkmenistan and Uzbekistan, as the former keeps expanding its irrigated acreage, thanks to the Karakum Canal, consuming larger amounts of the Amu Daria. Tension is altering relations between Uzbekistan and Tadjikistan, as the latter is complaining about the heavy prices Uzbekistan is demanding for the gas it delivers, whereas water in the Amu flows freely from Tadjikistan to Uzbekistan; Tadjikistan threatened to bill Uzbekistan for the resource, on the basis that the water comes from its territory. Tension also disrupted relations between Uzbekistan and Kirghizstan over the use of the Toktogul Dam water. Built during the Soviet era in mountainous and water-rich Kirghizstan, it was mainly meant to store water during the winter to provide water for the irrigated lands of the Ferghana Valley downstream, mainly in Uzbek territory. In January 2001, the Kirghiz government defaulted on its gas payments to Uzbekistan; the Uzbek government suspended gas deliveries that are mainly used for urban heating. Kirghizstan opened the Toktogul Dam sluices to produce electricity. The induced flood destroyed several dikes in the Uzbek Ferghana region. The Kirghiz pressure peaked when, after 15 days of letting the reservoir gradually empty to generate electricity, Kirghizstan notified the Uzbek government that they would not be able to guarantee water deliveries for the next growing season. This diplomatic skirmishing over energy deliveries bore fruit inasmuch as an agreement was signed by both countries for more cordial relations, recognizing the interests of Kirghizstan in using the waters of the Syr River—but it also underlined the acute tensions over the use of this very water.

It is not that the five Central Asian republics have not undertaken negotiations. As early as 1993, they signed the *General Agreement on the Crisis of the Aral Sea*, also signed by Russia, which provided water sharing cooperative mechanisms and technical cooperation to improve agricultural yields. Because of poor funding and lack of political will, the agreement was never enforced. As of 1995, the European Union and USAID stepped up their cooperation by creating large funds to help the functioning of institutions such as the *Interstate Council*

for Addressing the Aral Sea, or the Optimization of the Use of Water and Energy Resources in the Syr-Darya Basin project. So far, progress has been very slow.

THE TEMPTATION OF UNILATERALISM OR SOVIET STYLE MANAGEMENT

Soviet-style planning led to the idea that water was not particularly scarce; when it did become so, gigantic diversion of Siberian rivers, the Ob and the Ienissei, were designed by engineers to cope with the issue, only to be stopped in 1986 by Mikhail Gorbachev. It is all the more difficult for the Central Asian republics to reach an agreement as they lack financial resources to implement any conversion scheme in agriculture because cotton is a major export resource for the three downstream users. Besides, the economic recession that was triggered by the disintegration of cooperation within the former Soviet Union creates social unrest on which political opposition, particularly Islamic movements, capitalize to question the governmental authority, especially in Uzbekistan. In the latter, difficulties in reforming agriculture are compounded by the strong lobby of industrial cotton farmers.

Nationalist rhetoric, often associating water issues with ethnic tensions, developed in the region. This trend must be analyzed in the framework of tensions erupting between states as they close their respective borders and as governments have been charged with encouraging nationalist stances that help them stay in power in these troubled post-Soviet times. Besides, Uzbekistan is increasingly considered by its neighbors as pursuing a regional power policy (13).

Economic pressures and the slow progress in negotiations have encouraged several governments to act unilaterally to protect their interests. For instance, in 1994, Turkmenistan unilaterally resumed the extension of the Karakum Canal, thereby triggering strong protests from Uzbekistan. The annual volume diverted into the canal reportedly amounts to between 15 and 20 km³ (14). Similarly, in 1998, Kazakhstan succeeded in building a dike between the Little (or Northern) and the Large Aral, trying to keep the flow of the Syr Daria in the Little Aral to restore it gradually. The World Bank showed interest in the project. The Little Aral level grew by several meters and a few fish species have been reintroduced. The Kazakh government hopes that this project can help save the Little Aral, but it also means that the Large Aral is doomed, as any conservation project must rely only on the Amu Daria. Uzbekistan strongly protested against the Kazakh scheme. Kazakh officials replied that Uzbekistan has not shown any real desire to reduce its water consumption.

CONCLUSION

The Aral Sea is disappearing because of the development of unsustainable levels of agricultural demand for irrigation. Cotton is a major crop in Central Asia. The Soviet government vastly expanded this culture for social, political, and industrial reasons. To do so, water

development was planned regionally without considering the various federated republics.

When they became independent in 1991, the harsh economic recession and social unrest made any move to reform agriculture and implement water-saving measures difficult. Nationalist tensions, played upon by bureaucracies and governments, also played a role in stalling the negotiations that had begun as early as 1992. The Aral Sea keeps receding, and the temptation of unilateral decision-making remains strong in the five republics. Water withdrawals for irrigation have decreased since 1985, but they remain far too high to prevent the Aral Sea from disappearing.

Failure to reach an agreement could doom the disappearing sea, all the more if peace prevails in Afghanistan. A quarter of the drainage basin of the Amu Daria lies in that country. If peace is to return there, agriculture will resume, and with it, irrigation, thereby further reducing the available volume for Turkmenistan and Uzbekistan.

Acknowledgments

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LAKE BAIKAL—A TOUCHSTONE FOR GLOBAL CHANGE AND RIFT STUDIES

DEBORAH HUTCHINSON

STEVE COLMAN

U.S. Geological Survey

“The Lake Baikal rift system is a modern analogue for formation of ancient Atlantic-type continental margins. It tells us the first chapter in the story of how continents separate and ultimately develop into ocean basins like the Atlantic Ocean.”

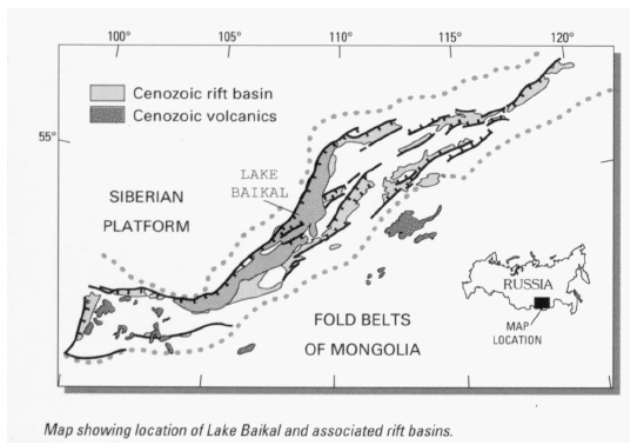
—Deborah Hutchinson

“Lake Baikal is a unique, nearly pristine environment for the study of global change. Nowhere else in the world can we go to study so long a record of such an important, but little known, part of the global climate system.”

—Steve Colman

THE SPECIAL ENVIRONMENTAL AND GEOLOGICAL SETTINGS OF LAKE BAIKAL PROVIDE UNPARALLELED OPPORTUNITIES FOR RESEARCH AND FOR INTER-NATIONAL COOPERATION

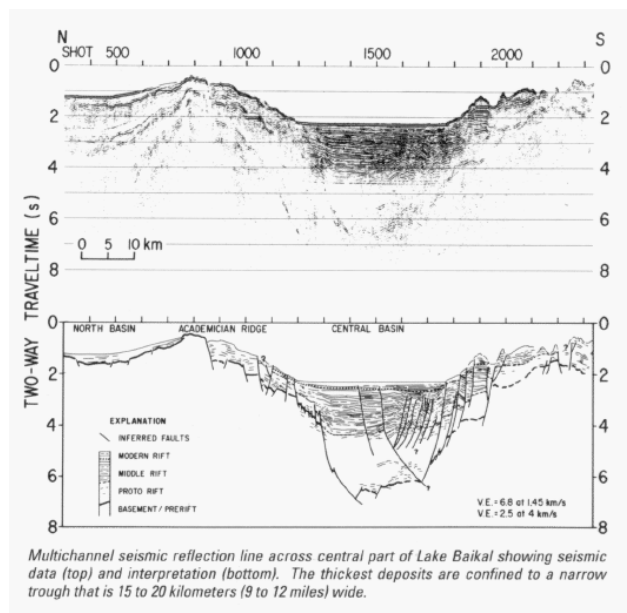
Lake Baikal is to Russia what the Grand Canyon is to the United States: a magnificent natural resource that instills national pride and awe. The Presidents of the United States and the Russian Federation recognized the uniqueness of Lake Baikal. In a recent Joint Statement, they affirm the need to conserve the environment of Lake Baikal and to use its potential for research in limnology, geology, and global climate change. The U.S.



Geological Survey's (USGS) Office of Energy and Marine Geology and Office of International Geology, supported by the Coastal Geology Program and the Global Change and Climate History Program, are involved in broad collaborative programs to study Lake Baikal with the **Russian Academy of Sciences** and with a number of American universities through the (U.S.) **National Science Foundation**.

UNDERSTANDING THE ORIGIN OF THE LAKE BAIKAL RIFT CONTRIBUTES TO UNDERSTANDING ONE OF THE FUNDAMENTAL PHENOMENA BY WHICH THE HISTORY OF THE EARTH IS RECONSTRUCTED

Continental rifts, like the Lake Baikal rift, and their end products, such as passive continental margins like the east coast of the United States, are ubiquitous in the Earth's geologic record. They contain information from which a significant amount of the Earth's history has been interpreted. Due to their high sedimentation rates, large rift lakes like Lake Baikal have great potential for providing high-resolution information about both tectonic and climatic change. Significantly, sedimentary deposits of continental rifts are also associated with many of the Earth's hydrocarbon and mineral deposits.



USGS AND RUSSIAN COOPERATIVE STUDIES HAVE BEGUN TO RESOLVE THE THREE-DIMENSIONAL GEOMETRY OF THE LAKE BAIKAL RIFT

Sediments of Lake Baikal reach thicknesses in excess of 7 kilometers (4 miles), and the rift floor is perhaps 8 to 9 kilometers (more than 5 miles) deep, making it one of the deepest active rifts on Earth. The shallowest sediments may contain the only known freshwater occurrence of natural gas hydrates. Maps of complex fault patterns and changing depositional environments provide the first opportunities to describe the development of the lake and to help explain its unique flora and fauna.



The RV Vereshchagin, a coring and high-resolution seismic vessel, docked on central Lake Baikal.

UNIQUE CHARACTERISTICS OF THE LAKE BAIKAL ENVIRONMENT COMBINE TO PRODUCE AN ESPECIALLY PROMISING SITE FOR STUDIES OF CLIMATE HISTORY

Lake Baikal is the largest freshwater lake on Earth containing 23,000 cubic kilometers of water, or roughly 20 percent of the world's total surface fresh water. It contains as much fresh water as the Great Lakes of North America combined. At over 1,600 meters (5250 feet), it is the deepest lake in the world, and at perhaps more than 25 million years old, the oldest as well. The water of Lake Baikal is so fresh that calcium carbonate does not survive in the fossil record. Despite the lake's great depth, its water is well-oxygenated throughout creating unique biological habitats.

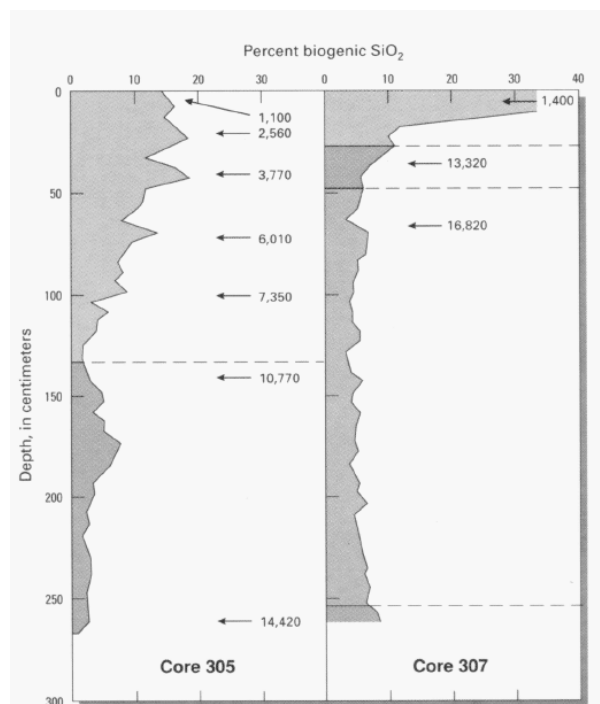
THE HIGH LATITUDE OF LAKE BAIKAL MAKES IT PARTICULARLY SENSITIVE TO CLIMATIC VARIATIONS

Climate variations, including those resulting from atmospheric accumulation of carbon dioxide, are more pronounced at higher latitudes. Although the lake contains a record of glaciation of surrounding mountains in its sediments, it is unique among large, high-latitude lakes in

that its sediments have not been scoured by overriding continental ice sheets.

UNITED STATES AND RUSSIAN STUDIES OF SEDIMENT CORES TAKEN FROM LAKE BAIKAL PROVIDE A DETAILED RECORD OF CLIMATIC VARIATION OVER THE PAST 250,000 YEARS

Much attention is focused on numerical models of climate change but there have been few means for reliably testing or modifying boundary conditions of general circulation models. Studies of sedimentary environments in Lake Baikal provide important opportunities to establish ground truth for general circulation models. Very little data exist for long-term climate change from continental interiors; most of the data record derives from the marine or maritime environments. Finally, studies of past environments contribute to understanding the extent to which human activity affects natural conditions in the lake.



Graph showing percentage of biogenic silica in two Lake Baikal cores. Since diatoms and algae are the most important microorganisms in the lake, biogenic silica is a measure of primary productivity. Radiocarbon ages in years are shown by arrows. First-order changes are caused by glacial (blue)-interglacial (red) cycles, with transitions shown in purple. Second-order fluctuations, such as the peak at 3,770 years, are due to shorter climatic events.

SEISMIC AND SEDIMENT CORE ANALYSES ARE USED TO FIX FUTURE DRILLING SITES IN LAKE BAIKAL

Ice-based drilling operations begun in early 1993 are providing longer (over 100 meters in length) cores of Baikal sediments. Analyses of these cores are expected to reveal the climatic, environmental, and geological history of the region as far back as 5 million years. Seismic data will be tied to cores and drill samples to estimate rates of



Ice-based drilling rig being tested on a small marginal lake in North Baikal. Photo by D. Williams.

climate change and to map the history of the lake and rift. Very deep drilling in Lake Baikal remains technologically challenging; therefore, the deepest deposits of the rift are not likely to be sampled soon. However, the potentially very long record of sedimentation in Lake Baikal provides unique opportunities to understand the Cenozoic climate history of the Earth and to describe how continents begin to break apart, giving rise to new ocean basins.

BASE FLOW

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Water may reach a river by flowing over the catchment surface, moving through the soil, or through groundwater. Based on the path taken by water, streamflow may be divided into surface flow, interflow, and base flow. The ASCE (1) defined base flow as the runoff that has reached the stream or river by passing first through the underlying aquifer, rather than by flowing directly on the ground surface. Thus, base flow is that portion of streamflow that is naturally and gradually withdrawn from groundwater storage or other delayed sources. The other names of base flow are groundwater flow, seepage flow, low flow, and fair weather flow.

Perennial streams depend on base flow for discharge between runoff producing events. The presence of base flow around the year indicates humid climate and a shallow water table that is hydraulically connected to the stream. Base flow is absent in (semi)arid climates and areas of deep groundwater. Of course, it is assumed that the streamflow is not influenced by storage or diversion.

FACTORS AFFECTING BASE FLOW

Base flow depends on precipitation, geologic conditions, and hydrogeologic controls that govern groundwater movement. Sources, such as lakes, marshes, snowpack, and channel bank, may also supply water for base flow. At a given time, more than one source may be supporting base flow. Climate influences recession through recharge and evapotranspiration.

Base flow contribution to streamflow varies widely according to the geologic nature of the water-table aquifer. A measure of the amount of water available from a water-table aquifer is the coefficient of storage, or specific yield. The lateral movement of groundwater is slower than vertical movement because the hydraulic gradient is smaller for lateral movement. For this reason, groundwater "mounds" and water tables beneath ridges may be at higher elevations than stream channels. This provides a gradient that directs base flow to the stream. The supply from groundwater to the channel will

continue as long as the necessary gradient is present. If no additional infiltration recharges an aquifer, the hydraulic gradient decreases as water moves to the stream from higher elevations. Consequently, less water will be discharged to the stream with time. This process is called base flow recession.

Bank storage may also contribute to base flow depending upon the position of the water table, riverbank material, soil characteristics near the riverbank, and the rise and fall of the stream. Usually, bank storage is largely depleted by the time the base flow peak occurs. Other water-table aquifers such as fractured rocks and solution cavity rock openings also contribute base flow to streams. The important parameters related to groundwater outflow are the total length of perennial streams, average basin slope, and drainage density. Singh provides a detailed discussion on base flow (2).

BASE FLOW RECESSION

Streamflow recession represents withdrawal of water from storage without inflow. It constitutes the falling limb of a hydrograph to the right of the point of contraflexure. When all surface runoff has passed, the flow in the stream consists of base flow. Figure 1 shows components of a typical hydrograph.

Base flow is at maximum immediately after a storm because precipitation has just recharged the water table. As water is withdrawn from an aquifer, groundwater supplies are diminished and base flow decreases. This process continues until all groundwater available for base flow is exhausted or additional recharge occurs.

Many different forms of base flow recession curves have been proposed. These are discussed in what follows.

Simple Exponential

The basic differential equation that governs the flow in an aquifer was presented by Boussinesq in 1877 (3). Boussinesq solved this equation by assuming that the Dupuit assumption of negligible vertical flow holds and the effect of capillarity above the water table can be neglected. This assumption leads to the solution,

$$Q_t = Q_0 \exp(-t/a) \quad (1)$$

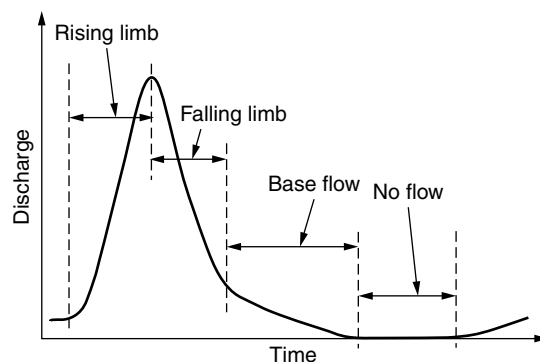


Figure 1. Sections of a streamflow hydrograph.

or its alternate forms

$$Q_t = Q_0 k_r^t \quad (2)$$

$$Q_t = Q_0 (10)^{-t/b} \quad (3)$$

where Q_0 is flow at time $t = 0$; Q_t is flow at time t ; and a , k , and b are constants. Parameter k_r is known as the recession constant or the depletion factor. It is a dimensionless quantity that depends on the unit of time selected (commonly, it is 1 day). It lies in the interval $[0,1]$ and is normally greater than 0.7. Constants a and b , known as storage delay factors, have the dimension of time. Constant b denotes the time required for the flow to decrease by a factor of 10 or one log cycle (see Fig. 2). Similarly, constant a denotes the time required for the flow to decrease by a factor equal to e , or one natural log cycle, as shown in Fig. 3. In other words, a is the time elapsed between any discharge Q and Q/e of the recession and is related to the 'half flow period' by (4)

$$a = -t_{0.5}/\ln(1/2) \quad (4)$$

where $t_{0.5}$ is the time required to halve the streamflow. Parameters a and k_r are related by

$$\begin{aligned} a &= -t/\ln k_r \\ k_r &= (10)^{-1/b} \end{aligned} \quad (5)$$

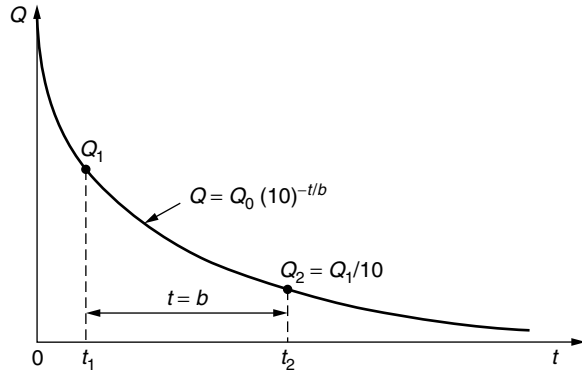


Figure 2. Base flow recession: determination of storage delay time b .

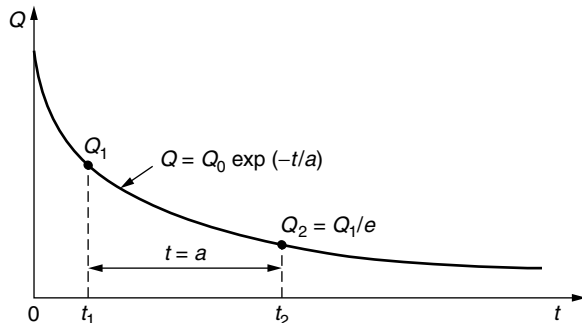


Figure 3. Base flow recession: determination of storage delay time a .

Horton (5) suggested a nonlinear recession equation, known as Horton's double exponential:

$$Q_t = Q_0 \exp(-a_2 t^m) \quad (6)$$

where a_2 and m are constants.

Singh (6) derived ideal base flow curves by solving the Boussinesq equation using a finite-difference equation for two cases: a shallow aquifer and fully penetrating stream and a deep aquifer and shallow-entrenched stream.

More on Exponential Recession

Equation 1 represents a first-order process or an exhaustion phenomenon expressed by

$$ds/dt = -Q, \quad Q(0) = Q_0 \quad (7)$$

and

$$s = aQ \quad (8)$$

where s is storage. The solution of Eqs. 7 and 8 is Eq. 1. The form of Eq. 1 does not change if discharges are replaced by volumes. The volume of flow during the time from $t - T$ to t is given by

$$\begin{aligned} V &= \int_{t-T}^t Q dt = \int_{t-T}^t Q_0 \exp(-t/a) dt \\ &= aQ_0 [\exp(T/a) - 1] \exp(-t/a) \end{aligned} \quad (9)$$

For a fixed time interval T , such as 1 day,

$$V = V_0 \exp(-t/a) \quad (10)$$

in which

$$V_0 = aQ_0 [\exp(T/a) - 1] \quad (11)$$

Thus, the recession constant can be estimated from Eq. 10. It may be preferable to use volumes instead of discharge because volume is less sensitive to errors than discharge.

Double Exponential

When base flow is plotted on semilog paper, the resulting curve is nonlinear which can be represented (5) by a double exponential of the form,

$$Q = Q_0 \exp(-mt^n) \quad (12)$$

where $\exp(-m) = k_r$ is the recession constant and n is a constant. Parameters m and n can be obtained either graphically or by using the method of least squares. By taking the log of both sides of Eq. 12 twice,

$$\ln(Q_0/Q) = mt^n \quad (13)$$

$$\ln[\ln(Q_0/Q)] = \ln(m) + n \ln(t) \quad (14)$$

Equation 14 represents a straight line.

Hyperbola

Another equation for a stream located on a horizontal impermeable lower boundary with an initial curvilinear water table and zero water-level elevation in the stream is of the form

$$Q = Q_0/(1 + ct)^2 \quad (15)$$

where c is a constant. This equation will hold where the groundwater storage is at its maximum after surface runoff stops or if no further precipitation or snowmelt occurred until the streamflow ceased. Parameter c can be estimated as

$$c = [(Q_0/Q)^{0.5} - 1]/t \quad (16)$$

The outflow from storage can be represented by a general non linear relation,

$$Q = kS^p \quad (17)$$

where S is storage and k and p are constants. The recession will be concave downward for $p > 1$ and upward for $p < 1$. Values of p greater than unity are typical of many catchments (4). The superposed exponential equation represents recession flow as a superposition of n exponential terms; each shows the response of a linear reservoir whose recession constant is c_i :

$$Q_t = \sum_{i=1}^n Q_{oi} \exp(-t/c_i) \quad (18)$$

Ice-Melt Recession

For snow and ice-melt conditions, base flow recession can be represented by (7)

$$Q = at^{-n} + b \quad (19)$$

where a , b , and n are constants. As time increases, Q asymptotically approaches a constant value b . This may typify base flow recession for permanent snow and ice. Constants a , b , and n can be determined either graphically or by using the method of least squares.

An exponential equation for base flow recession in watersheds that have permanent ice and snow is of the form (7),

$$Q = a + (Q_0 - a)k_r^t \quad (20)$$

where a and k_r are constants. This is similar to Eq. 19, where Q asymptotically approaches a constant value as t increases.

Base Flow Recession as Auto Regressive Processes

The base flow recession Eq. 2 can also be written as

$$Q_{t+1} = k_r Q_t \quad (21)$$

However, this equation is an approximation and contains errors due to a linearity assumption. The input

data also contain some errors and, therefore, Eq. 22 can be written as

$$Q_{t+1} = k_r Q_t + \varepsilon_{t+1} \quad (22)$$

where ε_t are independent, normally distributed errors that have zero mean and a constant variance. This equation represents a first-order autoregressive process, denoted by AR(1), where k_r is the autoregressive parameter. If the errors are additive in log-space, Eq. 23 becomes

$$Q_t = k_r Q_{t-1} e^{\varepsilon_t} \quad (23)$$

Vogel and Kroll (8) noted that the error structure of Eq. 23 is more representative of actual streamflow records. Base flow recession can also be interpreted as an AR(2) process:

$$Q_t = (\varphi_1 Q_{t-1} + \varphi_2 Q_{t-2}) \varepsilon_t \quad (24)$$

Another way to look at recession is as an analogy with a linear reservoir where the storage-outflow relationship is assumed linear:

$$S = kQ \quad (25)$$

where S is storage, Q is outflow, and k is a constant whose dimension is time. The continuity equation for a reservoir is

$$I - Q = dS/dt \quad (26)$$

Assuming that the inflow is zero and letting $Q(t_0) = Q_0$ (the outflow at $t = t_0$) yields

$$Q_t = Q_0 \exp[-(t - t_0)/k] \quad (27)$$

Equation 2 can be written in discrete form as

$$Q_{j\Delta t} = Q_0 k_r^j = Q_{(j-1)\Delta t} k_r \quad (28)$$

where $t = j\Delta t$. Comparing Eq. 27 with Eq. 28,

$$k = -\Delta t / \ln k_r \quad (29)$$

Substituting from Eq. 27 in Eq. 26, letting $I = 0$, and integrating,

$$S_t = -Q_0 / \ln k_r \quad (30)$$

DETERMINATION OF BASE FLOW RECESSION CONSTANTS

The recession constant k_r or parameters a and b can be determined in a number of ways.

Graphical Method

Constants a and b can be determined by constructing Figs. 2 and 3 and choosing any two values of discharge that satisfy the condition specified there. Note that these constants vary with the choice of discharges. Therefore, it is appropriate to obtain a number of values of a and b

and then take an average. Alternatively, one can take the logarithm of Eqs. 1–3,

$$\ln(Q) = \ln(Q_0) - t/a \quad (31)$$

$$\ln(Q) = \ln(Q_0) + t \ln k_r \quad (32)$$

$$\log(Q) = \log(Q_0) - t/b \quad (33)$$

Base flow discharge Q can be plotted against time on a semilog paper using any of Eqs. 31–33. The recession parameters can be determined by fitting a straight line to the plotted data. Alternately, one can also plot Q_{t-1} versus Q_t on a simple graph. For example, if Eq. 2 is valid, this plot will result in a straight line passing through the origin. Its slope will be k_r . Similar interpretations can be advanced for Eqs. 2 and 3. For example, Eq. 1 can be recast as

$$Q_t = Q_{t-1} \exp(-T/a) \quad (34)$$

where Q_t is the discharge at time t and T is the time interval between Q_{t-1} and Q_t . The plot of Q_{t-1} versus Q_t should give a straight line that has the slope $\exp(-T/a)$. Obviously, the slope depends on the time interval, which is commonly 1 day.

Least Squares Method

James and Thompson (9) determined the recession constants by using the least squares method. Any of Eqs. 1–3 can be used with this method. For illustration, Eq. 2 is recast as

$$Q_t = k_r Q_{t-1} \quad (35)$$

The recession constant can be estimated by minimizing the sum of the squares of the differences between the observed and computed values of discharge:

$$R = \sum_{t=1}^N (Q_t - k_r Q_{t-1})^2 \quad (36)$$

where N is the number of discharge values and Q_t corresponds to the observed values. To estimate k_r , the derivative of R with respect to k_r is set equal to zero to yield

$$k_r = \sum_{t=1}^N Q_t Q_{t-1} / \sum_{t=1}^N Q_{t-1}^2 \quad (37)$$

Statistical Methods

The recession constants can be estimated by the method of moments. Only one parameter is to be estimated, so the first moment about the origin will suffice. Again, any of Eqs. 1–3 can be used. To illustrate, consider Eq. 1. Let $M_1(Q)$ be the first moment of Q about the origin. Then,

$$M_1(Q) = \int_0^\infty t Q_0 \exp(-t/a) dt / \int_0^\infty Q_0 \exp(-t/a) dt = a \quad (38)$$

Thus, recession constant a for a given base flow sequence is equal to the first moment of that sequence about its origin.

If Eqs. 22 or 24 are treated as regression equations, the least-squares method can be used to estimate the recession constant. For the AR(1) model given by Eq. 22, the ordinary least squares estimate of k_r is

$$k_r = \frac{\sum_{t=1}^{n-1} Q_{t+1} Q_t}{\sum_{t=1}^{n-1} Q_t^2} \quad (39)$$

where n is the total number of consecutive observations for a site.

While comparing the various techniques to estimate recession constants, Vogel and Kroll (8) found that the recession constant obtained by treating the base flow as an AR(2) process that has an additive residual in log-space (e.g., Eq. 24) led to a regional regression model of low-flow statistics that has the highest R^2 , lowest standard error, and highest t -ratios of model parameter estimates.

MASTER BASE FLOW RECESSION CURVE

The master base flow recession curve is a composite recession curve that represents mean recession behavior. Of course, the information about recession variability is lost in the process. This curve can be constructed in several ways (7); the most popular are experimentation and the correlation method.

Experimentation involves observing the discharge at a number of time intervals encompassing the entire dry-weather period. These are then plotted against time on semilog paper, and a best fit straight line is drawn through the plotted points. The resulting curve is a recession curve.

The correlation method involves plotting the discharge for the initial day of each segment on log-log paper, starting with the largest discharge, against discharge T time units later (where $T = 1/2, 1, 2, 5$, etc.). Plotting is continued for as many segments as possible until a good correlation is established for drawing a line. The points above the line represent surface flow and those below it represent recession.

Algorithms for Selecting Recession

The beginning of a recession segment can be a fixed discharge or a variable. A fixed value restricts the domain of recession analysis to flows less than that value. The beginning may also be specified as a certain time after the rainfall or after the peak of the hydrograph. Sometimes, some initial values are ignored to minimize the influence of surface flow, and some values near the end may be ignored to obviate the influence of the next storm.

Vogel and Kroll (8) describe an automatic hydrograph recession selection algorithm to separate the recession part from the discharge data. In this, a recession begins when a 3-day moving average begins to decrease and ends

when the same begins to increase. Recessions longer than 10 days or more were identified by them.

Applications of Base Flow Recession

A major practical application of base flow recession is forecasting low flows in a river for drought management, planning drinking water supply, maintaining water quality in the river, or ecological planning. Other applications of base flow analysis are determining the relation between hydrologic and geologic parameters of a drainage basin, evaluating the effect of agricultural practices, locating suitable areas for induced infiltration, controlling withdrawal of groundwater for irrigation during low flow periods, and determining storage requirements to maintain minimum flow in rivers.

BASE FLOW SEPARATION

Base flow separation, also called hydrographic analysis, is the process of separating surface runoff from base flow. Even though such separation is somewhat arbitrary and subjective, it is useful in many analyses. Several techniques have been developed for base flow separation. Some popular techniques are described below.

Area Method

The area method of base flow separation is based on a nonlinear relation between time and area (10):

$$N = bA^{0.2} \quad (40)$$

where A is the drainage-basin area in km^2 , b is a coefficient equal to 0.8, and N is the time in days from the hydrograph peak (see Fig. 4). This equation is not suitable for smaller watersheds. It generally gives a longer time base. For example, if $A = 1000 \text{ km}^2$, then $N = 3.18$ days, if rainfall occurs for 6 hours, its effect will be felt for more than 3 days.

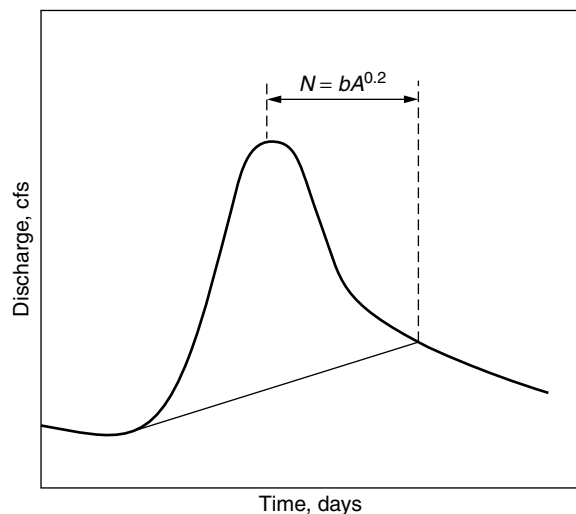


Figure 4. Base flow separation based on area.

It is convenient to draw a separation line directly from the chosen groundwater discharge on the receding limb to the point under the hydrograph peak. Although this linear separation does not represent the true boundary between direct runoff and groundwater runoff, the error may be acceptable in most cases.

Three-Component Separation

Three-component separation involves separating surface runoff, interflow, and base flow. The method, developed by Barnes (11), is based on Equation 2 and is illustrated in Fig. 5. First, streamflow recession is plotted on semilogarithmic paper. In Fig. 5, the groundwater recession plots approximately as a straight line, where $K_r = 0.992$. By extending this straight line under the hydrograph to the point directly under the point of inflection E and to B on line AB, points B and J are connected arbitrarily by a straight line. The area under the hydrograph above BJH is considered direct flow and that area below BJH is considered groundwater flow. The direct runoff is replotted, and a straight line IL where $K_r = 0.966$ is fitted and extended to point I directly under inflection point E and to the beginning point M. The line MIL divides the replotted hydrograph into surface runoff on top and interflow below.

Singh and Stall Method

Although Barnes' method has been used for quite some time, it is subjective and has many shortcomings, as pointed out by Kulandaiswamy and Seetharaman (12). Several methods have been developed to overcome these drawbacks and remove subjectivity. The method developed by Singh and Stall (13) is described here.

1. Select single peak hydrographs that are not closely followed by another stream rise.
2. Estimate the storage delay factor K_b and the range of base flow Q_b .

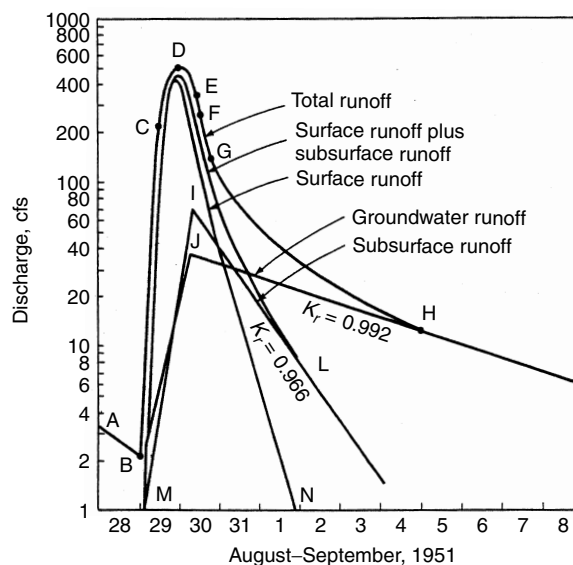


Figure 5. Three-component hydrograph separation.

3. Using an average value of K_b for the desired range of Q_b , begin at a suitable value of Q_b where the hydrograph shape suggests the end of flow from the surface runoff stored in the channel system, and compute daily values of Q_b by progressing backward in time to within 1 or 2 days of the inflection point on the hydrograph recession. Draw the straight-line base flow hydrograph.
4. Subtract Q_b from Q_T , the total discharge and plot the $(Q_T - Q_b)$ hydrograph on a semilog paper. Draw a best fit straight line.
5. Draw a few $(Q_T - Q_b)$ hydrographs by using slightly higher and lower values of K_b . The value of K_b that makes the $(Q_T - Q_b)$ plot nearest to a straight line is adopted. The straight line is denoted as the Q_s line, and the number of days for Q_s to drop in amount by one log cycle is the value of K_s in days.
6. Compute $(Q_T - Q_s)$ by starting a few days after the inflection point on the Q_T hydrograph, and obtain the base flow hydrograph. Note the time in days T_b by which the base flow peak lags behind the hydrograph peak.
7. Analyze a number of hydrographs to obtain sets of values of K_b , K_s , T_b , and the range of Q_b for which a particular K_b is suitable.
8. Draw base flow recession curves for both growing and dormant seasons.
9. Compute values of Q_s using different values of K_s . Add Q_b and Q_s to get Q_T . Draw traces of total hydrograph recessions by starting from various points on the base flow recession curve. The Q_T recessions will define the average limits within which actual Q_T recessions will lie.

Base Flow Separation using Digital Filters

Graphical methods of base flow separation are inconvenient for processing a long streamflow record. Digital filters are helpful in such situations. Basically, streamflow is partitioned into two components:

$$Q(i) = Q_l(i) + Q_h(i) \quad (41)$$

where $Q_l(i)$ is the low-frequency component and $Q_h(i)$ is the high-frequency component. The first can be interpreted as base flow and the second as overland and quick interflow.

A filter employing the two-parameter algorithm of Boughton (14) assumes that base flow at the current time is a weighted average of the direct runoff and the base flow at the previous time interval. Let $Q_b(i)$ and $Q(i)$ be the base flow and total streamflow for the time interval i . Then,

$$Q_b(i) = [p/(1+C)]Q_b(i-1) + [C/(1+C)]Q(i) \quad (42)$$

subject to

$$Q_b(i) \leq Q(i)$$

where p and C are the parameters of the model; p can be interpreted as a recession constant. The two-parameter

algorithm was found the most satisfactory by Chapman (15). In this case, the parameters have less effect on the BFI (base flow index which is the ratio of the long-term ratio of base flow to total stream flow) than other models. Furey and Gupta (16) developed a filter from a physical point of view.

Chemical and Isotopic Techniques

Chemical and isotopic tracers provide another means to separate a hydrograph into components. An additional advantage of this method is that one can also identify the spatial source of water. The technique has been applied in many studies (17,18). However, its use in base flow separation requires elaborate field work and laboratory analysis that is time-consuming and expensive.

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RIVER BASINS

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RIVER BASIN DEFINITION AND DELINEATION

A river basin (also called drainage basin, watershed, or catchment) is the area that provides runoff to a given section of a stream and is separated from adjacent basins by a divide. All surface runoff produced by precipitation that falls within the area is discharged through the lowest point in the divide.

Basins are delineated on topographical maps, integrated if necessary with aerial photographs or field survey; a line is drawn that encloses all drainage lines and depressions in the basin and passes through the highest points. It is assumed that the movement of groundwater follows the surface divides, but in areas characterized by a particular geologic structure, it is possible that the phreatic divide does coincide with the topographic divide; large amounts of water could be transported below the surface from one basin to another. In this case, the groundwater divide needs to be determined by hydrogeologic prospection.

A river basin is generally considered the fundamental unit for geomorphologic and hydrological studies; it can be treated as an open system that receives energy from the climate (input) and loses energy by evapotranspiration to the atmosphere and by stream flow and sediment yield through the basin mouth (output) that determine storage changes in groundwater and soil moisture (Fig. 1).

A river basin may be described by a Geographic Information System (GIS) combined with a digital elevation model (DEM). A GIS is an electronic system of maps connected to tables of data that describe the features on the maps; a DEM represents a spatial distribution of elevations. The DEM can be structured by a square-grid network, a contour-based network, or a triangulated irregular network (TIN) (1). In a raster-based GIS, the basin is subdivided into uniform grid cells; to each of these cells, numerous attributes relative to basin characteristics can be attached. The GIS system is a useful tool for automating much of the geometric data acquisition necessary for hydrologic analyses (2). A raster-based GIS is well suited to the application of hydrologic rainfall–runoff models based on distributed parameters. Alternatively, the GIS procedure allows computing average basin parameters for the entire watershed that are necessary as input in lumped parameter hydrologic models.

BASIN CHARACTERISTICS

A river basin is composed of several subsystems: the drainage basin itself in its areal extent, and the channel system, in which by a further subdivision, we can distinguish the channel cross section, the channel reach, and the channel network. It is important to describe the river basin in quantitative terms to understand the processes and to analyze the interrelationships existing among these characteristics.

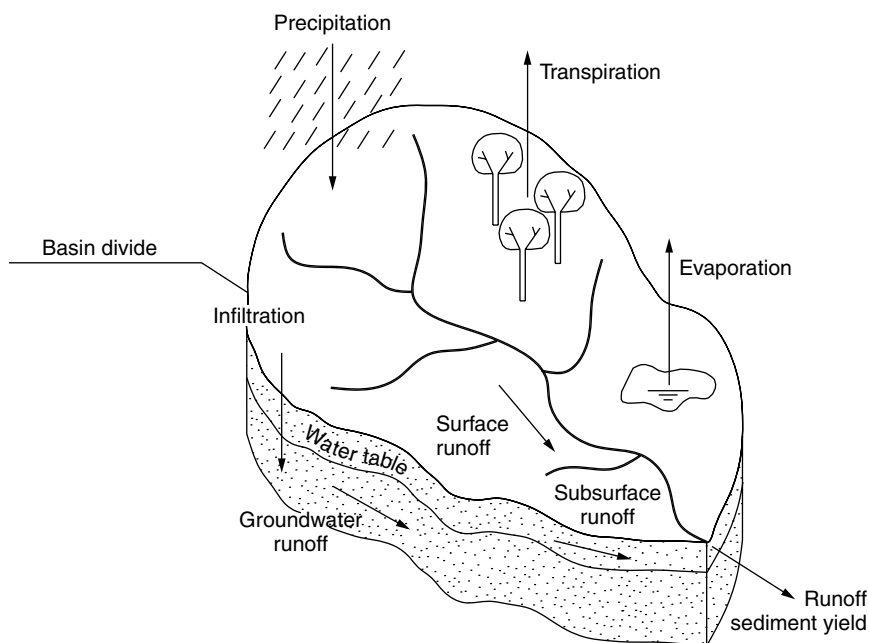


Figure 1. River basin.

Drainage Basin

A river basin's characteristics can be subdivided into topographic, rock and soil, vegetative and land use, and climatic characteristics.

Topographic Characteristics. Topographic characteristics directly influence the transport of water and sediment within a basin. Topographic attributes may be described by index of area, length, shape, and relief.

Basin Area. The area of a drainage basin is the area enclosed by the curve obtained by projecting the basin divide on a horizontal plane. This represents its most important physical parameter because it is correlated with almost every other characteristic, but for this reason it is not always easy to interpret its significance.

Empirical relations between stream discharge Q and basin area A assume the form,

$$Q = aA^b \quad (1)$$

where Q represents some index of stream flow such as the mean annual flood (3). The value of exponent b generally varies from 0.5 to 1.0. Stream flow per unit area is inversely proportional to area because intense precipitation tends to occur on a restricted area. Sediment yield per unit area in many regions, it has been found, is inversely proportional to basin area. Erosion is predominant in small basins, whereas in large basins, there are, in proportion, more locations for temporary sediment storage.

Basin Length. Basin length is usually defined as the length of the mainstream measured along the main channel from the outlet to the basin divide. Length is closely related to area, and a large number of rivers seem to satisfy the relation expressed by Hack:

$$L = 1.4A^{0.6} \quad (2)$$

where L is the stream length in miles and A is the area in square miles (4). The exponent 0.6 suggests that geometric similarity is not preserved as a river basin increases in area because A/L^2 decreases and the basin tends to elongate, becoming longer and narrower.

Basin Shape. The basin shape normally tends to a pear-shaped ovoid, but many substantial deviations from this shape due to geologic constraints have been observed. Several parameters have been proposed to describe basin shape. The form factor F is defined as what is F ?

$$R_f = \frac{A}{L^2} \quad (3)$$

where A is the drainage basin area and L the basin length.

The basin circularity compares the basin area with the area of a circle that has the same perimeter p :

$$R_c = \frac{4\pi A}{p^2} \quad (4)$$

The basin elongation is defined as the ratio of the diameter of a circle that has the same area as the basin to the basin length:

$$R_e = \frac{2\sqrt{A}}{L\sqrt{\pi}} \quad (5)$$

R_e assumes values from 0.6 to 0.8 in regions of high relief, and in areas of low relief, values near one have been observed. The usefulness of these shape factors has been analyzed, and the elongation ratio has proved to be the parameter best correlated with hydrology (5). The basin shape affects the stream-flow hydrograph, in particular, the lag time and the time of rise.

Relief. The hypsometric area–altitude analysis expresses the relation between basin area and elevation. It is obtained by plotting the area above a certain elevation versus the elevation itself. The shape of the *hypsometric curve* gives an indication of the basin's geologic evolution any stage in the erosion cycle. The hypsometric curve is useful if hydrologic variables such as precipitation or evaporation, which vary with altitude, are studied.

The *mean basin elevation* can be obtained by integrating of the area–elevation curve and dividing the result by the drainage area; the median elevation is defined as the elevation that corresponds to 50% of the drainage area.

The *maximum basin relief* is the elevation difference between the highest point on the basin perimeter and the basin outlet.

The *basin slope* is a major factor in the overland flow process, and several methods have been developed for determining its average value because of the variation in land–surface slope throughout the basin. In the grid-square method, a grid of uniformly spaced lines is established over a map of the basin, and the slope of a short segment of line normal to the contours is determined at each grid intersection. The basin slope is then calculated as the mean of the resulting local slopes. In the random-coordinate method, the procedure is the same except that the points where the local slope is evaluated are randomly located over the basin (4). Basin slope can also be determined by measuring the total length L_{tot} along contours at contour intervals Δz and calculating the average value:

$$i = \frac{\Delta z L_{tot}}{A} \quad (6)$$

Computations by this formula are time-consuming, but by introducing a DEM, they become more feasible. However, using a DEM, it is also possible to calculate the slope of each individual square or triangle in a raster or TIN model, respectively, and to use these values to compute the average basin slope.

The peak discharge per unit of basin area is strongly correlated with the mean basin slope. In basins that have a high relief ratio, the lag time and time of hydrograph rise are shorter, and the peak discharge rate is higher (Fig. 2A). Sediment erosion that takes place mainly during peak stream flow is also correlated with basin relief.

Rock and Soil Characteristics. Rock and soil characteristics determine the rate of rainfall infiltration into the soil

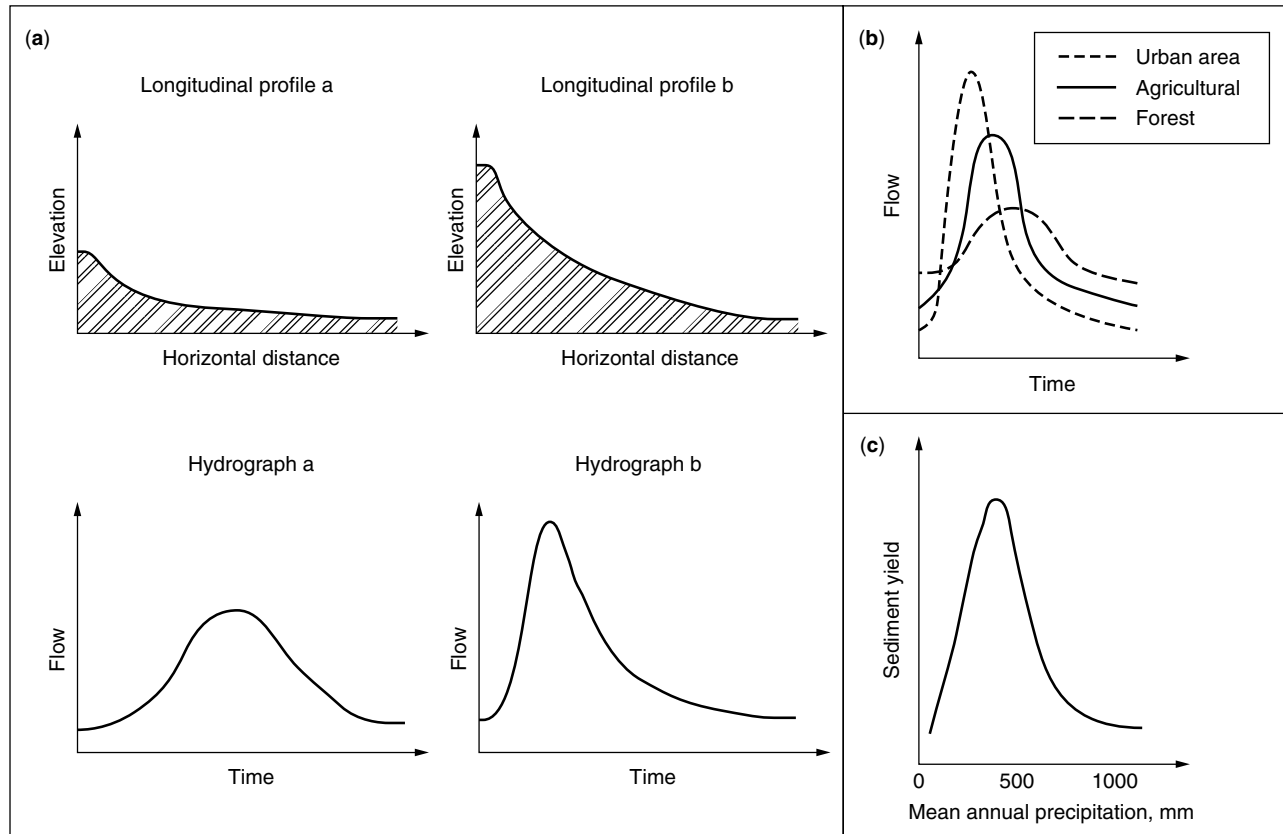


Figure 2. Influence of relief (a) and vegetative cover (b) on runoff and influence of climatic characteristics (c) on sediment yield.

and thus affect runoff. The character of the rock beneath the basin with regard to these processes can be expressed by porosity and permeability. *Porosity* is the percentage volume of voids in a material and is a measure of the amount of water that can be retained by a deposit; *permeability* provides an index of the ability of a material to transmit water and is a function of soil texture, grain size, density of grains, and particle shape. The Soil Conservation Service has classified most soils into four hydrologic soil groups A, B, C, and D, dependent on their infiltration rate (6):

- group A: deep, well-drained sand;
- group B: sandy loam;
- group C: clay loam;
- group D: clay soils of a high swelling potential.

The geology of a basin affects the type and the amount of material available for erosion and transport. The extent of removal and transport of sediment is influenced by the physical and chemical properties of the rock. The vulnerability of soils to weathering can be evaluated by several indexes of soil erodibility which depend upon particle size, moisture content, permeability, dispersion properties of the soil and shear resistance.

Vegetation and Land Use Characteristics. Vegetative cover is a significant factor influencing drainage basin dynamics because it affects the amount of net precipitation

that reaches the ground surface through interception, evapotranspiration, and infiltration, and also the water and sediment rate produced by the basin.

Interception losses are given by the sum of precipitation retained on vegetative surfaces such as leaves, stems, and tree trunks. The amount of water captured by vegetation depends on the amount of leaf cover; the type, age and density of vegetative cover; seasonal variations; the amount, duration, and intensity of precipitation; and antecedent moisture conditions.

Vegetative cover also determines losses by evapotranspiration, which is the combination of evaporation from plant surfaces and transpiration of water from plants. Evapotranspiration is related to plant characteristics and environmental conditions, such as atmospheric pressure and moisture, air temperature, wind speed, soil moisture, and light intensity.

The effect of vegetative cover is also displayed by an increase in soil infiltration capacity due to the resistance afforded by vegetation, which slows down overland flow and thus allows the water to have more time to enter the soil surface and flow to the plant roots which make the soil more previous to infiltrating water. The type and density of vegetation, the presence of plant litter on the soil surface, and the depth and density of roots may influence infiltration dynamics.

The vegetative cover influences stream-flow runoff in a basin (3). As the vegetation extent decreases, passing from forested areas to agricultural land and further to

urban areas, the range of flows between peak and low flows becomes wider, and the peak discharge is amplified and occurs earlier (Fig. 2B).

Sediment production by erosion depends on land use character, especially cover density. Vegetal cover has a protective influence against erosion processes. A dense cover such as grass reduces the flow velocity and operates simultaneously as a protection covering the underlying soil. Sediment production is maximum from cultivated land, decreases in pasture land, and is minimum in forest and wild land.

Climatic Characteristics. Climatic characteristics govern the amount of water received at the surface. Sediment yield is broadly correlated with mean annual precipitation (7). Maximum values occur in semiarid areas, where the proportion of precipitation available for surface runoff and therefore for erosion is greater. In more arid and more humid areas, sediment production decreases because of the reduced runoff potential and the increase in vegetative cover, respectively (Fig. 2C).

Channel System

Channel Cross Section. The shape and size of alluvial channel cross sections are strongly correlated with stream discharge. Numerous relationships that express the variation in channel characteristics, such as top width, mean depth, mean velocity, slope, with discharge at a particular section and between cross sections have been developed (9). As discharge increases downstream, surface

width and mean depth tend to increase, whereas slope decreases. Downstream variations in channel form are influenced by local factors such as type and quantity of sediment load, rock type, superficial deposits, and vegetative cover. Thus, quantitative relationships between hydraulic geometry and discharge may be applied only locally in the same region where the data from which the equations are derived have been obtained, for example, to estimate stream flow from channel measurements in a site where no stream-flow records are available.

Channel Reach

Channel Slope. The slope of the principal drainage channel can be estimated by several methods (Fig. 3A). The simplest method consists of dividing the difference in elevation between the source and the mouth by the length of the channel (Definition 1). The slope can also be determined by calculating the slope of a line drawn on the channel profile through the lowest point such that the area under it equals the area under the profile curve (Definition 2). Channel slope affects flow velocity, travel time, and sediment transport capacity, so it influences water and sediment routing through the basin.

Channel Patterns. River channel patterns have been classified into meandering, braided, and straight (Fig. 3B).

A channel is meandering where it assumes the form of a number of loops or bends. The *sinuosity* is the ratio of channel length to valley length and has an average value of 1.5. Meander wavelength and amplitude range

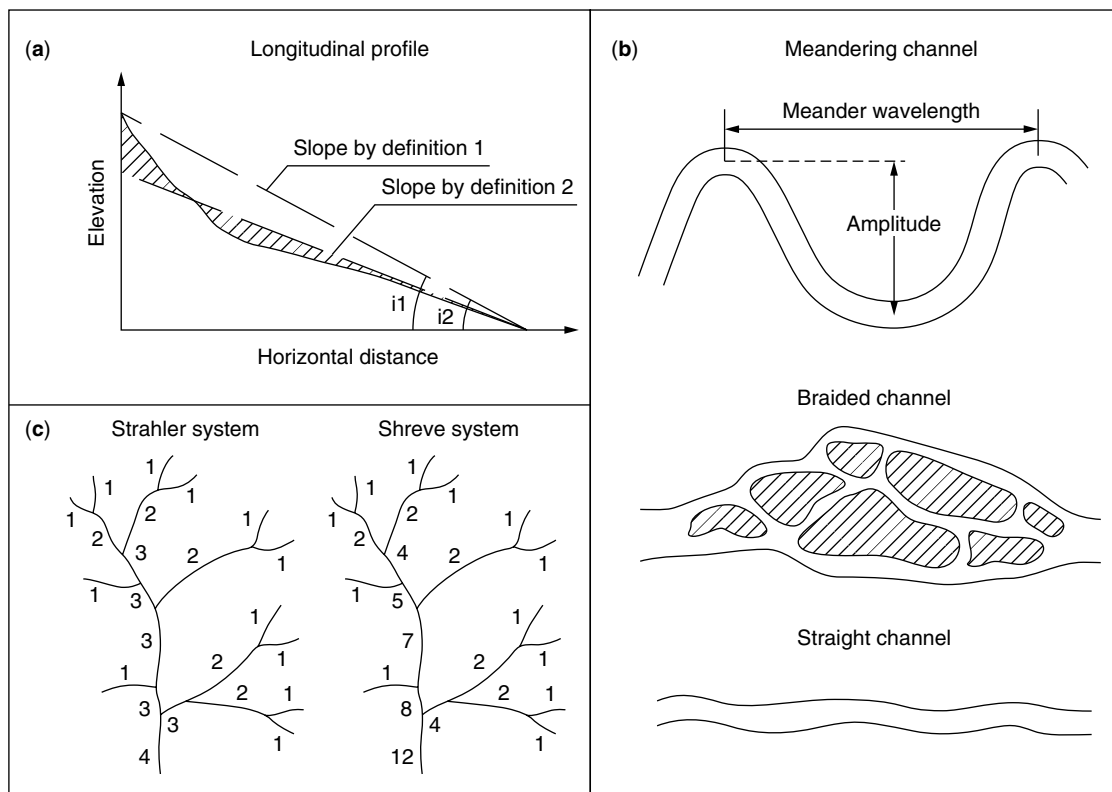


Figure 3. Channel system: (a) Channel slope definition; (b) channel pattern description; (c) stream order designation.

from 7 to 10 and from 3 to 20 times the channel width, respectively (7). Meander wavelength has proved to be strongly correlated with discharge. A channel is braided where the stream bifurcates into two or more intertwined channels separated by islands or bars. Braided channels are characterized by wide, shallow beds of coarse material. The factors that contribute to the development of braided channels are essentially bed-load availability, irregularity of stream flow, intense precipitation which causes high peak discharges, and high slopes (3). Perfectly straight channels are seldom found in nature, so the definition of straight channel has been extended to a channel whose sinuosity is less than 1.25.

Channel Network

Drainage Density. The density of a channel network is the total length of channels per unit area. Values of drainage density observed range from 1 to 248 km/km². High density is observed in basins that have weak and impermeable rocks, sparse vegetation, and a steep relief. Network density depends on climatic characteristics, and it varies with mean annual precipitation in the same way as sediment yield because high sediment delivery corresponds to a well-developed channel network. Drainage density is inversely related to precipitation-effectiveness Thornthwaite's index P-E which adjusts monthly precipitation by a function of monthly average daily temperature to account for evapotranspiration losses (4). This relation shows the effectiveness of vegetative cover in erosion control. Other basin characteristics such as lithology and land use, and in particular, rock permeability, influence drainage density because they affect the infiltration-runoff rate. Network density is an important factor in the formation of flow. As it increases, peak discharge per unit area increases, and time-to-peak decreases, so that a basin that has a high drainage density is characterized by an efficient drainage system and rapid hydrologic responses.

Stream Order. A channel network may be described by designation of a stream order using with one of the systems developed by Strahler and Shreve to determine the position of a stream in the hierarchy of tributaries (Fig. 3C). In the Strahler system, all fingertip tributaries have order one; two first-order channels joining produce a second-order segment, and so on; if two channels of different order join, the resultant link has the order of the tributary of higher order. In the Shreve system, all fingertip tributaries have magnitude one, and each downstream channel segment has a magnitude equal to the sum of all the first-magnitude segments that are tributary to it. The basin order is a measure of the branching within the basin and is related to the size of channel network and to the amount of stream flow. Stream order is very sensitive to map scale which should always be specified together with the method of ordering used. The Strahler method is the most widely used method today, especially combined with the laws of drainage composition, whereas the Shreve method gives a description of a drainage network closer to physical reality with regard to stream discharge.

Horton's laws of drainage composition relate stream numbers, stream lengths, and drainage areas to stream order by simple geometric relationships (4). The bifurcation ratio R_B is defined by

$$R_B = \frac{N_u}{N_{u+1}} \quad (7)$$

where N_u is the number of streams of a given order and N_{u+1} is the number of the next higher order. Bifurcation ratios tend to be constant in a given basin and assume characteristic values between three and five. The law of stream numbers is expressed by

$$N_u = R_B^{k-u} \quad (8)$$

where k is the highest order within the basin. Similarly, the laws of stream lengths and drainage areas indicate a geometric progression of order:

$$\bar{L}_u = \bar{L}_1 R_L^{u-1}, \bar{A}_u = \bar{A}_1 R_A^{u-1}, R_L = \frac{\bar{L}_u}{\bar{L}_{u-1}}, R_A = \frac{\bar{A}_u}{\bar{A}_{u-1}} \quad (9)$$

where L_u is the mean length of streams of order u , A_u is the mean area of basins of order u , R_L is the length ratio, and R_A is the area ratio, whose ranges of values are 1.5–3.5 and 3–6, respectively.

An alternative model has been proposed by Shreve using the probabilistic-topological approach. It assumes that in the absence of climatic and geologic constraints, channel networks evolve at random; all topologically distinct channel networks (TDCN) of a given magnitude are equally probable (5). Shreve used the term "link" for stream segments between nodes (sources, junctions, or outlet) and distinguished exterior links, which extend from a source to the first junction, from interior links, which connect two successive junctions or the last junction with the outlet. Furthermore, he assumed that exterior and interior link lengths are random variables whose separate probability distributions are independent of position within the basin. Shreve showed that the most probable network in a random population closely conforms to Horton's law of stream numbers.

More recently a new approach has been developed for analyzing drainage network composition, which shows that river basins have a self-similar organization, that is, the same shape is found at another place in another size. This property can be interpreted by Mandelbrot's fractal theory which introduces fractals as irregular geometric shapes that have identical structure on all scales (8).

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RIVER BASIN PLANNING AND COORDINATION

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A river basin is a functional entity where water is the integrating factor. What happens in one location will impact any downstream location. Sometimes there are upstream impacts as well, for instance, backwater effects upstream of a dam. Thus, water management must be coordinated and concerted to result in desired outcomes in different parts of a drainage basin. To this end, there is a need for effective communication across institutional and disciplinary boundaries.

As stated in Agenda 21 [a wide-ranging blueprint for action to achieve sustainable development worldwide]. Adopted by United Nations Conference on Environment and Development (UNCED), Rio de Janeiro, June 3–14, 1992., freshwater is a unitary resource. Long-term development of global freshwater requires holistic management of resources and recognition of the interconnectedness of the elements related to freshwater and freshwater quality. Failures in these respects have resulted so that there are few regions of the world that are still exempt from problems of loss of potential sources of freshwater supply, degraded water quality, and pollution of surface and groundwater sources.

In chapter 18 of the Agenda, current wisdom concerning water management is summarized, and it is stated that freshwater resources are an essential component of the earth's hydrosphere and an indispensable part of all terrestrial ecosystems. Water is needed in all aspects of life, and the general objective of water management is to make certain that adequate supplies of water of good quality are maintained for the entire population of this planet, while preserving the hydrologic, biological, and chemical functions of ecosystems.

It is pointed out that the extent to which water resources development contributes to economic productivity and social well-being is not usually appreciated. Thus, a spreading scarcity, gradual destruction, and aggravated pollution of freshwater is allowed to go on without much attention from responsible agents/institutions. A major problem is the fragmentation of responsibilities for water

resources development among sectoral agencies such as those concerned with water supply and sanitation, agriculture, industry, urban development, hydropower generation, inland fisheries, transportation, recreation, and low- and flatland management.

In addition, different, but inherently linked aspects of catchment management, such as water quality, water quantity, and the processes of erosion and deposition, are frequently managed by different institutions. Often, there is also a division of responsibility for surface water issues and groundwater issues, respectively. These aspects need to be coordinated if we are to create the synergy needed to achieve the desired water status. Coordination is needed at both the political and technical levels.

An important starting point is that water should be managed at the river basin level and be seen as an integral part of the ecosystem, a natural resource, and a social and economic good, whose quantity and quality determine the nature of its use.

Efficient coordination is essential for the dynamic, interactive, iterative, and multisectoral approach called for. The holistic management of freshwater as a finite and vulnerable resource urgently calls for integrating sectoral water plans and programs within the framework of national economic and social policy. Various levels of complexity should be recognized when coordinating activities by different agents or organizations. Simplistic solutions should be avoided.

Four basic requirements can be identified:

1. management of the whole resource;
2. clear links between catchment functions;
3. emphasis on prevention and mitigation rather than responses;
4. adoption of multifunctional win-win solutions.

Water management should take into account long-term planning needs as well as those with narrower horizons. The long-term need should be related to the principle of sustainability; the more short-term objectives would be focused on the prevention and mitigation of water-related hazards.

The International Conference on Freshwater in Bonn, 2001, stated that water management arrangements should take account of climate variability and expand the capacity to identify trends, manage risks, and adapt to hazards such as floods and droughts. Anticipation and prevention are more effective and less expensive than having to react to emergencies. Early warning systems should become an integral part of water resources development and planning. It was also stated that systematic efforts are needed to revive and learn from traditional and indigenous technologies (for example, rainwater harvesting) around the world.

Closer links should be established between development and disaster management systems. Exposure to flood risks should be minimized through wetland and watershed restoration, better land use planning, and improved drainage. The greater fluctuation in resource availability associated with presumed climatic changes is causing concern, as many regions experience increasingly severe

flooding and/or drought problems. The solutions to some of these problems can be sought locally, whereas others have to be dealt with in international forums.

Thus, the scale of issues must be understood to allow for a prioritized approach. Often, issues are perceived differently by different groups of people, and myths about their severity are legion. For instance, within basins embracing several nation states, the attitudes of those states in the headwaters is typically different from the attitudes of the states in the lower reaches of the river. Headwater states do not wish to be held to guarantees of water quantity or of water quality demanded by their downstream neighbors. Downstream countries are, very naturally, concerned about the prospects of any upstream dam construction which may affect the regime or total quantity of flow, especially by major diversions of water into or out of the basin (1).

With regard to existing uncertainty and lack of precise knowledge, it is essential to monitor continuously and evaluate what happens to water quality and quantity at key points and to make sure that the information gained is made available to those who need it, when they need it.

Strategies and programs need to be tied to indicators that enable actors to assess progress. As we are swamped by a wealth of information, a major problem is that we have difficulties in discriminating between information, useful knowledge, and science. Therefore, it is essential to have few but relevant indicators of high explanatory value. Jiménez-Beltrán (2) claims that efficient indicators should

1. show development over time and be policy relevant (there should be an explanation why a specific indicator has been developed);
2. be few in number and people should get used to their presentation and understand the message;
3. be closely linked to objectives to become efficient tools in decision-making.

In the United Kingdom, local Agenda 21 efforts show that indicators work best when they are developed in participatory ways. In communities of all kinds, "everyday experts" are getting involved in the monitoring and evaluation process, and results are starting to change policy (3).

On a more conceptual level, Brown (4) suggests that to be efficient, indicators should be

1. holistic—measuring if the catchment is worked with as a whole;
2. cooperative—measuring the degree of working together with shared responsibilities;
3. composite—measuring the degree of inclusion of the full range of diverse elements and the degree to which elements have been arranged to fit together;
4. coordinated—assessing the mechanisms for continuing cooperation; and
5. long term—assessing the durability of relationships and processes and assessing the progress toward a shared vision.

Basic to all activities is a baseline assessment. We have to know where we start from before we can decide how to go to the desired conditions. Such assessment, including the identification of potential sources of freshwater supply, comprises the determination of sources, extent, dependability, and quality of water resources and of the human activities that affect those resources. It is a prerequisite for evaluating the possibilities of water resources development. There is, however, growing concern that, in a time when more precise and reliable information is needed, hydrologic services and related bodies are less able to provide this information, especially concerning groundwater quantity and quality.

In most places, there is a need to strengthen the institutional arrangements for water assessment. This is not just about efficient collection of data, but also about processing, storage, retrieval, and dissemination to users of information about the quality and quantity of available water resources at the level of catchments and groundwater aquifers.

Prior to such strengthening, it is necessary to prepare catalogues of the water resources information collected and/or held by government services, the private sector, educational institutes, consultants, local water-use associations, and others. It is essential to build on what exists by strengthening and coordinating the different bodies involved in the collection, storage, and analysis of the relevant data.

There is a need to strengthen the technical support by installing additional observation networks, developing systems for data storage, and systems for data retrieval. However, strengthening of the human resource base in numbers and skills seems to be equally important.

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BIOACCUMULATION

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Bioaccumulation is a process by which a chemical from the environment accumulates in an organism's body over time. Bioaccumulation occurs when there is a concentration increase of the chemical in the tissues of an organism.

Bioaccumulation results from mass equilibrium where the uptake of a chemical into the body of an organism exceeds the elimination of the chemical from the body. As such, a chemical that is eliminated immediately after uptake will not be bioaccumulated in an organism.

Uptake of a chemical means entrance of the chemical into an organism primarily from water or soil, directly or through consumption of foods that contain the chemical. A chemical can be taken up through roots or leaf surfaces in plants and by breathing, ingesting, or absorbing through the body surface in animals. Uptake of a chemical in plant cells can occur against its electrochemical potential gradient through the process of active transport. For example, selenate (SeO_4^{2-}) and arsenate (AsO_4^{2-}) are taken up via sulfate and phosphate transporters in the root plasma membrane, respectively.

The elimination of a chemical from an organism includes excretion and/or metabolism of the chemical. Elimination is the primary process by which an organism can reduce its chemical burden. Excretion is the removal of toxicants or excess chemicals from the body of plants or animals. A chemical taken up into the body of an animal can be eliminated by urination and defecation, whereas salt-tolerant halophytes can excrete sodium and other ions from their leaf surfaces.

A chemical can also be eliminated from an organism by transforming the original chemical into a new chemical, a process termed metabolism. Inorganic selenate can be taken up and biologically reduced or methylated into organic selenium compounds that can be volatilized into the atmosphere (1). Metabolism can also break down a large organic compound into small molecules or eventually mineralize the compounds into water, CO_2 , and nutrients. Metabolism of an organic compound by an organism is termed biodegradation.

An organism's ability to take up or eliminate a chemical varies among species and depends on physiochemical properties of the chemical. Chemicals that are water-soluble will be taken up and readily eliminated by an organism. Persistent organic compounds are generally not water-soluble and are difficult for organisms to metabolize or degrade. These compounds tend to have a greater potential to accumulate in fat or lipid tissues.

Bioaccumulation can sometimes substantially increase the concentration of a chemical in the tissues of an organism. When the concentration of a chemical becomes higher in an organism than in the environment (e.g., water, soil, or air), the bioaccumulation process is specifically termed bioconcentration. The extent of bioconcentration can be expressed by the bioconcentration factor ($\text{BF} = C_{\text{org}}/C_{\text{env}}$), which is a ratio of the concentration of a chemical in an organism (C_{org}) to the concentration of the same chemical in the living environment (C_{env}). If a plant species can accumulate an exceptionally high level of a metal (e.g., about 1% dry weight for zinc and manganese or 0.1% for copper and nickel in shoots) without having harmful effects, the species is termed a metal hyperaccumulator. *Astragalus bisulcatus* is a selenium hyperaccumulator that can accumulate about 0.5% (dry weight) of selenium in shoots.

A chemical that is bioconcentrated/bioaccumulated is not homogeneously distributed in the body of an organism. Distribution of an accumulated chemical varies significantly among tissues/organs, species, and chemicals. For instance, fat-soluble chemicals (e.g., polychlorinated biphenyls—PCBs) accumulate primarily in fat, cadmium in kidneys, and mercury in livers. Such organ-specific accumulation of a chemical may impose a potential hazard to an organism because the chemical can reach a critical body burden in those target organs. This can even occur in organisms that are exposed only to low levels of a toxicant in the environment.

Plant roots generally accumulate higher concentrations of metals (e.g., lead and manganese) than shoots. The distribution of a chemical in a plant can be described by its translocation factor ($\text{TF} = C_{\text{shoot}}/C_{\text{root}}$), the ratio of the concentration of the chemical in shoots (C_{shoot}) to the concentration in roots (C_{root}). A larger translocation factor means that a greater proportion of the chemical is accumulated in shoots compared to that in the roots. Similarly, the ratio of the shoot concentration (C_{shoot}) to the soil concentration (C_{soil}) of a chemical is termed the phytoextraction coefficient ($\text{FC} = C_{\text{shoot}}/C_{\text{soil}}$). Phytoextraction is one of the phytoremediation approaches using plants to remove toxicants from contaminated waters and soils. Plants with high phytoextraction coefficients can remove a large amount of pollutant by harvesting the chemical-laden shoots at contaminated sites.

One of the important environmental concerns associated with bioaccumulation is biomagnification, whereby the concentration of a chemical increases at each higher trophic level through a food chain. Biomagnification can result in a concentration increase of two to three orders of magnitude between two trophic levels. Therefore, due to biomagnification through the entire food chain, the concentration of a chemical in top predators may reach tens of thousands times the concentration in the water column. For example, through a water–plankton/algae–fish food chain at the Kesterson reservoir in Central California, selenium in fish was biomagnified 35,000 times from selenium-contaminated drainage water. Very high concentrations of selenium result in reproductive deformities and death of fish and waterfowl (2).

The term bioaccumulation is sometimes used to refer to the total amount of a chemical accumulated in the body of an organism. The amount of a chemical accumulated in an organ or tissue depends on both the chemical concentration and the actual biomass in which the concentration is determined. For example, a higher bioconcentration of selenium in *Astragalus bisulcatus* does not necessarily mean that a greater total amount of selenium from the environment can be accumulated in the plant because the species grows slowly and the total biomass production is small. To overcome such biological limitations, scientists are currently applying new biotechnology to combine the genome of a tolerant, slow-growing, selenium hyperaccumulator, such as *Astragalus*, with that of a less tolerant, but fast growing nonhyperaccumulator, for example, *Brassica juncea*, to develop a somatic hybrid plant that could increase the phytoremediation potential (3).

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BIOTIC INTEGRITY INDEX TO EVALUATE WATER RESOURCE INTEGRITY IN FRESHWATER SYSTEMS

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Many varied and complex environmental problems, particularly row crop and grazing agriculture, urbanization, highway building, timber harvesting, and water projects such as dams, water withdrawals, and stream channel modifications have seriously affected the health of freshwater resources in many parts of the world. The health of aquatic resources can be measured in many ways. Among them, biotic (or biological) integrity is a concept most commonly used by the public, biologists, resource managers, and policy makers to measure the status of aquatic systems.

Biotic integrity is “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region.” This concept was first proposed by Frey (1) and later applied by Karr and Dudley (2) in discussing an ecological perspective on water quality goals.

Aquatic systems of high biotic integrity have a biological community in which composition, structure, and function have not been seriously altered by human activities. Such systems can withstand or rapidly recover from some perturbations imposed by natural environmental processes and survive many major disruptions induced by humans. Aquatic systems that lack integrity are often degraded and when further perturbed by natural or human-induced events are likely to change rapidly to an even more undesirable status.

Biotic integrity should not be confused with biotic diversity. Biotic diversity (or biodiversity) refers to “the variety and variability among living organisms and the ecological complexes in which they occur” (3). More thorough definitions include multiple organization levels

(e.g., genes, species, and ecosystems) as fundamental components of biodiversity to distinguish it from the much simpler concept of species diversity (4).

Fundamental differences exist between biotic integrity and diversity. Biotic integrity refers to a system’s wholeness, including the presence of all appropriate elements and occurrence of all processes at appropriate rates, whereas diversity is a collective property of system elements (5). Biotic integrity is also associated with conditions under which the biotic communities evolved, but diversity may not necessarily measure such an aspect. For example, introducing exotic species or genes from distant populations may increase local diversity, but it reduces biotic integrity. Some aquatic management activities for increasing harvestable products or services of economic value may not necessarily correlate with biotic integrity. A large amount of harvestable products in some systems may indicate low biotic integrity (6).

FACTORS AFFECTING BIOTIC INTEGRITY

Biotic communities in aquatic systems have evolved over millions of years. The environmental conditions under which biotic communities evolved can be classified into five major groups (6). The energy source group includes the type, amount, and particle size of organic materials entering an aquatic system from the surrounding terrestrial zones and internal primary production. The water quality group includes temperature, turbidity, dissolved oxygen, pH, nutrients, heavy metals, and natural and synthetic inorganic or organic chemicals. The habitat group includes substrate type, water depth, water current velocity, habitat complexity, and reproduction and hiding places. The flow regime group includes water volume and temporal distribution of floods and low flows. The biotic interactions group includes competition, predation, disease, and parasitism. Regional climate, landscape topography, geology, soil type, and watershed land-cover type largely determine the factors within these five groups. Without human disturbance, the factors in aquatic systems and their watershed conditions are in dynamic equilibrium, which operates under natural weather cycles. As such, many of these factors have natural patterns of temporal and spatial variation.

When watershed lands are converted from forest, prairie, or wetland to agriculture and urbanization, the dynamic equilibrium between the elements in an aquatic system and its watershed is broken, causing degradation in aquatic resources. Degradation is intensified by aggressive farming practices such as overapplying fertilizers, pesticides, and herbicides to improve crop yields, concentrating livestock in high densities to increase production efficiency, and channelizing streams and draining wetlands to expand agricultural acreage. Urban development further degrades aquatic resources by increasing toxicants, nutrients, and storm-water runoff, in turn, causing more frequent and severe flooding, accelerated channel erosion, and an altered stream channel and substrate composition. Agricultural and urban land-use impacts cause major changes in aquatic communities and thus, biotic integrity within the system.

WHY USE BIOTIC INTEGRITY TO MEASURE WATER RESOURCE HEALTH

The combined influences of human-induced stressors are often difficult to detect, particularly across diffuse urban areas or agricultural landscapes. For example, water quality impacts are often associated with increased surface runoff from rainstorms or snowmelt. These episodic events can be assessed only by continuous monitoring, which is expensive and time-consuming. Even from continuous monitoring, runoff volumes and characteristics tend to be highly variable, and long time-series of data are required to evaluate trends. If changes in concentrations of a pollutant are detected, its effects on the health of the aquatic ecosystem often remain unclear.

Chemical and physical criteria are insufficient to protect water resources. The earliest anthropogenic threats to water resources were often associated with human health, especially disease-causing organisms and oxygen-demand wastes. Early emphasis was on controlling these contaminants in urban areas, where effluents exceeded the natural waste assimilating capacities of water, by using chemical and physical indicators as primary regulatory tools to protect water resources; this has eliminated or greatly reduced the known-source problems. However, water resource quality and quantity continued to decline despite massive governmental regulatory efforts. Nonpoint-source pollution inputs, those originating from diffuse areas such as farm fields and parking lots, were not extensively reduced by chemical and physical criteria alone. For example, in 1986, nonpoint-source pollution affected 65% of the impaired stream length, 76% of the impaired lake area, and 45% of the impaired estuary area in the United States (7).

Biotic assemblages represent the end point of the combined influences of human-induced perturbations (8). As such, biotic indicators can provide measures of water resource health, although diffuse disturbances within a watershed, from agriculture or urbanization, have many physical and chemical effects on a waterbody. Direct measures of biotic assemblages, such as fish, aquatic insects, and algae, are cost-effective and ecologically relevant ways to assess human impacts on aquatic resources. Because the biotic assemblages consist of a variety of species that have different life histories, sensitivities to degradation, and functions in the ecosystem, they respond to a range of human disturbances. A few appropriate samples of the assemblage can provide unique insight into the condition of the aquatic system and the causes of degradation. Biotic assemblages are accurate and easily measured indicators of the overall quality or health of water resources (9).

BIOTIC INDICATORS

Numerous assemblage-level indicators are available for assessing human impacts on aquatic systems. The most commonly used and effective indicators for water resource health can be grouped into three categories: tolerance, taxa richness and diversity, and reproducing and feeding ecology. Most indicators were developed empirically, based

on observed responses to a gradient of degradation, rather than experimental tests of sensitivity to a specific stressor. This approach is appropriate because human activities invariably have multiple interactive impacts on aquatic ecosystems that are often diffuse and cumulative. Although different types of indicators tend to be most sensitive to different environmental impacts, the best indicators are sensitive to most or all stressors that are typical of watershed disturbance.

Tolerance measures are based on the documented relative sensitivity of particular taxa to disturbance. The presence or abundance of taxa that have a known degree of sensitivity to a particular stress is a measure of the degree of that stress within the system. For example, Hilsenhoff (10) developed an aquatic macroinvertebrate index to assess organic pollution or the addition of excessive nutrients to a waterbody, which is a common consequence of agriculture and urbanization. Each taxon was assigned a tolerance value based on its sensitivity to organic pollution. The index was the weighted average, based on relative abundance, of the tolerance values of all of the taxa collected in a semiquantitative sample. If most of the taxa and individuals present were sensitive to organic pollution, then the index score was good, but if the assemblages were dominated by taxa tolerant of organic pollution, then the index score was poor. By their nature, tolerance measures tend to be relatively narrow in their sensitivity, but they have been strong indicators of water resource health.

Taxa richness is based upon the premise that the number of taxa is related to the amount and type of human disturbance. Diversity measures are based on the premise that both the number of taxa and evenness of the distribution of individuals among taxa are related to the amount and type of stress. Usually, richness and diversity are inversely proportional to stress, although this may not hold for "cold-water" streams where fish assemblages are dominated by salmonids. In warm-water streams, strong negative relations have been reported between watershed disturbance and both fish taxa richness and diversity, where the number of native, sensitive taxa decreased as stress increased. High natural variability in diversity scores has caused diversity measures to lose popularity, but taxa richness continues to be one of the most reliable and accepted indicators worldwide.

Reproducing and feeding ecology measures categorize organisms into groups that use similar reproduction and food resources. Multiple classification systems exist for classifying organisms by reproducing and feeding ecology, but most are based on combinations of what the animals eat, where they forage, how they acquire food, and what substrate they spawn on. Documenting the relative abundance of organisms that rely on each reproducing or feeding class reflects the habitat and food web conditions in which they live. Such measures also reflect the availability of essential life-cycle elements, energy flow, and nutrient dynamics, as well as the balance of these components between a water body and its surrounding terrestrial environment. Energy production and flow are difficult to measure directly, but biota provide surrogate measures.

MULTIMETRIC INDEXES OF BIOTIC INTEGRITY

A multimetric index of biotic integrity (IBI) combines a variety of different indicators, termed metrics, including tolerance, taxa richness, and reproducing and feeding ecology, into a single index that reflects structural, compositional, and functional attributes of an assemblage (9,11). By incorporating several different metrics, IBIs are sensitive to a wide range of human disturbances. Potential metrics are selected based on knowledge of aquatic systems, life history of the organisms, literature reviews, and historical data. The candidate metrics are evaluated and eliminated if they are not robust or show little relationship to human disturbances for the particular region and waterbody type of interest. The ability to interpret individual metrics is retained, even though the metrics are combined into a multimetric index.

An IBI must be regionally calibrated to take into account and correct for natural variation in assemblages owing to the biogeography or ecological conditions of the waterbodies. For example, it is well established that in the absence of human perturbations, larger streams tend to have more fish species than smaller streams and certain river basins have richer fish faunas than other basins because of geomorphic history. It is initially difficult to discern which factors are influencing the biotic metrics, natural environmental variability or human disturbance. Waterbodies that suffer relatively little human disturbance yet are influenced by the same regional set of natural environmental conditions (e.g., geology, climate, and waterbody type or size) are used as references to provide a baseline for comparison with waterbodies of unknown condition. Well-designed IBIs take natural differences into account and have metrics based on standards that are specific to the region and type of waterbodies for which they are intended.

Individual metric scores and final IBI values are relative to the set of reference waterbodies for a particular waterbody type and region. Test waterbody conditions are compared with a set of reference conditions to indicate their deviation from optimal conditions. Ranked scores, for example 5, 3, and 1, are assigned to each metric according to whether its value approximates, deviates somewhat from, or deviates strongly from the values expected at the reference condition. The overall index value is the sum of the ranked scores from all of the individual metrics. The highest possible score indicates a waterbody that is comparable to those that have the lowest stress within that region, and those of reduced quality have lower values. Based on the final value, one can qualitatively classify a water body as having good, fair, or poor health.

Fish Index of Biotic Integrity

Fish IBI is used worldwide and is especially popular in the United States. The IBI was first developed by Karr (12) for stream fish in the central United States. This version of IBI consists of 12 metrics that reflect basic structural and functional characteristics of fish assemblages: (1) number of native species; (2) number of darter species; (3) number of sucker species; (4) number of sunfish (excluding green sunfish) species; (5) number of individuals in sample;

(6) number of intolerant species; (7) percent green sunfish; (8) percent hybrid individuals; (9) percent omnivores; (10) percent insectivorous cyprinids; (11) percent top carnivores; and (12) percent disease, tumors, fin damage, and other anomalies.

Because IBI is regionally specific, many versions of fish IBIs have been developed to meet the needs in several parts of the United States, Canada, France, Mexico, Australia, Belgium, Guinea, India, Namibia, and Venezuela. Many versions have also been adapted to specific types of waterbodies, including headwater streams, wadeable streams, nonwadeable rivers, cold-water streams, cool-water streams, lakes, wetlands, and estuaries.

During the past 20 years, fish IBIs have been broadly used to document impacts of watershed land uses on the health of streams, rivers, and lakes. One example of the studies that comprehensively evaluated the impacts of urbanization on streams using IBI was conducted in Wisconsin, United States (13). In this study, fish communities in 54 warm-water and 38 cold-water streams were sampled by electrofishing stream segments that were 35 times the mean wetted stream width or at least 100 m. Fish were sampled between late May and early August, when low stream flows facilitated sampling effectiveness and large-scale seasonal fish movements were unlikely. During sampling, efforts were made to collect all fish observed, and all captured fish were identified, counted, and then returned to the stream alive. The collected data were then entered into a computerized database, and an IBI value was calculated for each stream. The cold-water and warm-water IBI versions used here were specifically developed for the study region. The scores for both versions ranged from 0–100; higher scores indicated better stream health. One simple way to evaluate impacts of urban land use on stream health is to plot percentages of watershed urban land use against the IBI scores (Fig. 1). Some streams had very healthy conditions, and others did not. These plots indicated that at low percentages of urban land use, the stream conditions were varied. At low levels of urbanization (less than 12%), urban influences were weaker than other stressors, such as agriculture. As the proportion of urban land use increased in the watersheds, the fish IBI values decreased sharply. When land use exceeded certain levels, the stream health was consistently poor. Such a relation indicated that as urban land use increased, urbanization dominated over all other land uses and played the strongest role in influencing stream health.

Karr (12) noted several advantages of using fish IBIs to measure stream health: (1) life history information is extensive for most fish species; (2) fish communities generally include a range of species that represent a variety of trophic levels and include foods of both aquatic and terrestrial origins; (3) fish are relatively easy to identify; (4) both acute and cumulative effects can be evaluated; and (5) the general public can relate conditions of fish community to water resource health. The disadvantages of using fish as indicators include (1) seasonal and diel fish movements, (2) intensive field sampling effort, and (3) modifying the IBI for specific geographic regions and waterbody types.

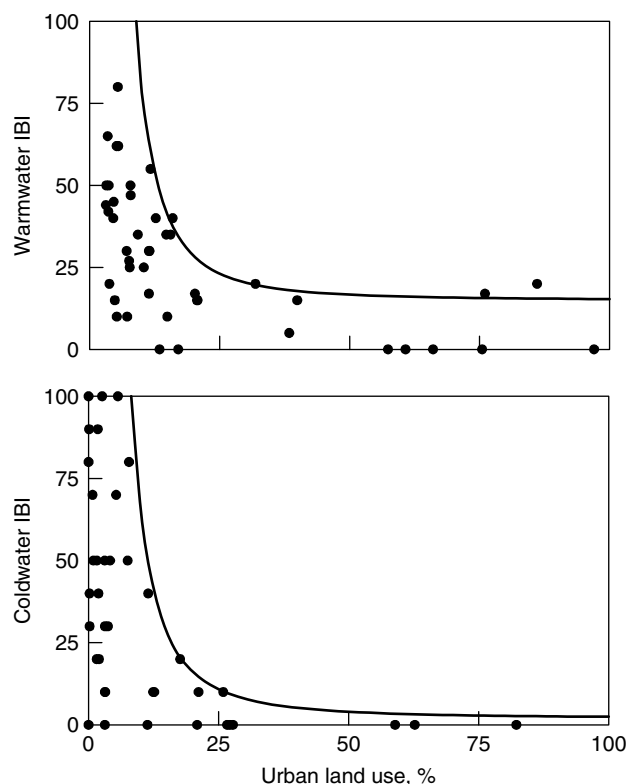


Figure 1. Relationships between watershed urban land use and fish index of biotic integrity for 54 warm-water and 38 cold-water streams in Minnesota and Wisconsin, United States. The land uses in the study watersheds are predominated by either agriculture or urban. At low level urbanization, urban influences of fish are weaker than other stressors, such as agriculture. As percentages of urban land in a watershed increase, fish IBI values decrease sharply and when urban land exceeds a certain level, the stream health is consistently poor.

Macroinvertebrate Index of Biotic Integrity

The macroinvertebrate IBI concept was extended from fish IBI and has been widely used in streams, rivers, and wetlands in North America. One of the early macroinvertebrate IBIs was developed for streams in Ohio, United States (14). This IBI, called the Invertebrate Community Index (ICI), consisted of 10 compositional and structural community metrics: (1) total number of taxa; (2) number of mayfly taxa; (3) number of caddisfly taxa; (4) number of dipteran taxa; (5) % mayfly composition; (6) percent caddisfly composition; (7) tribe tanytarsini midge composition; (8) percent other dipteran and noninsect composition; (9) percent tolerant organisms; and (10) number of Ephemeroptera–Plecoptera–Trichoptera taxa. Metric scoring criteria were developed through a quantitative calibration process in which reference values were plotted against a log-transformed watershed area.

Macroinvertebrate IBIs have also been broadly used to document human impacts on the health of streams, rivers, and wetlands in North America and other parts of the world. One example of such a study was also conducted in Wisconsin, United States, in which stream macroinvertebrate IBI values were empirically linked with watershed and local stressors (15). In this

study, streams were stratified by region to account for natural environmental variation. Standardized methods were used to kick-sample macroinvertebrates from 397 independent streams and identify the samples in the laboratory. Kick-sampling is a method for dislodging macroinvertebrates by kicking the substrate and letting the current wash the organisms into a net; it is typically done in riffle or run habitats that have coarse substrates.

Quantifying human disturbance entailed using standardized methods to characterize the watershed land cover and assess the local in-stream and adjacent terrestrial (riparian) habitat upstream of the sampling point. Land cover categories included urban, row crop agriculture, forage crop agriculture, wetland, forest, and open water. Local habitat characterizations included assessment of the riparian vegetation, bank erodability, livestock grazing, macroinvertebrate food sources, channel morphology, and streambed. These local and watershed measures of human disturbance were summed to give an overall environmental condition value to each site. Likewise, several macroinvertebrate metrics were combined into a multi-metric IBI to give an overall biotic integrity value to each site. A plot from 209 streams in the central–southeast region of Wisconsin shows how values of environmental condition were related to macroinvertebrate IBI values (Fig. 2). Streams of excellent environmental condition typically had relatively high proportions of forests, wetlands, and open water and a low percentage of urban land cover on the watershed scale. On the local scale, these excellent streams had a variety of food sources, relatively undisturbed riparian conditions, and a heterogeneous habitat structure, including riffles, meanders, and woody debris. The macroinvertebrate assemblages

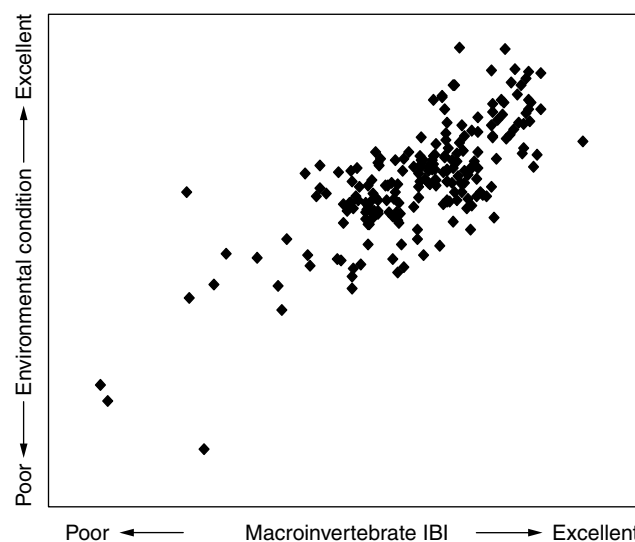


Figure 2. Macroinvertebrate index of biotic integrity tailored to small streams in central and southeast Wisconsin, United States ($n = 209$). The poorest streams had high proportions of urbanization or agriculture in their watersheds and local habitat stress. Macroinvertebrates in the poorest streams were tolerant to organic pollution, had no individuals from the relatively sensitive insect orders Ephemeroptera, Plecoptera, and Trichoptera, and had low taxa richness.

at these excellent streams had high species richness, were intolerant of organic pollution, and were comprised predominantly of intolerant taxa from the insect orders Ephemeroptera, Plecoptera, and Trichoptera, as opposed to Diptera. The empirical model indicated that extensive agriculture and urbanization impacted the biological community strongly and significantly.

There are several advantages of using macroinvertebrates to evaluate the health of aquatic systems. Macroinvertebrate assemblages consist of a variety of species whose different life histories are sensitive to a multitude of degradation types and play an important functional role in stream ecosystems. The assemblages can respond to a range of human-induced effects on streams. Macroinvertebrate assemblages represent the end point of the combined influences of hydrology, physical habitat, and water quality and it has been shown, respond predictably to these factors within specific geographical regions (9). Similar to the fish IBI, the disadvantages of using the macroinvertebrate IBI to measure the health of aquatic systems include (1) seasonal variation in their relative abundance, (2) intensive efforts for field sampling and taxa identification, and (3) modifying the IBI for particular geographic regions and waterbody types.

Periphyton Index of Biotic Integrity

The multimetric periphyton IBI was developed to measure the health of aquatic systems. Periphyton are algae attached to hard substrates like the rocks that comprise a streambed. The states of Montana and Kentucky were among the first to develop a multimetric periphyton IBI in the United States. Bahls' (16) periphyton IBI for Montana streams included three metrics based on soft-bodied taxa and four metrics based on diatom taxa. Metrics for soft-bodied taxa included dominant phylum, indicator taxa, and number of genera. Diatom metrics included the Shannon–Wiener diversity index, a pollution index, a siltation index, and a similarity index for comparison with a reference condition. Kentucky's stream periphyton IBI consisted of diatom species richness, species diversity, percent community similarity to reference sites, a pollution tolerance index, and percent sensitive species (17).

Periphyton respond to a variety of pollutants and can be used to diagnose the probable causes of health impairment in aquatic systems. Some periphyton can be sensitive to pollutants that other organisms tolerate relatively well, partly because periphyton cannot avoid pollutants due to their sedentary nature. Periphyton occur in most aquatic habitats, and typically have greater taxonomic richness than fish and macroinvertebrate. Although sampling periphyton is relatively easy, identification must be in a laboratory and it is labor-intensive. Typically, periphyton have rapid reproduction rates and short life cycles, and thus, they respond quickly to perturbation. However, this quick response to stress appears to increase their natural variation, which can make the detection of changes resulting from human perturbation more difficult.

PREDICTIVE MODELING

The River Invertebrate Prediction and Classification System (RIVPACS) and Australian River Assessment System (AusRivAs) are predictive models that quantify river health as the degree to which a waterbody supports the biota that would be expected there in the absence of human disturbance. These models were initially developed in the United Kingdom (18) and later modified for Australia (19), other parts of Europe, and North America.

In the process of assessing aquatic health, an empirical model of reference waterbodies is created that incorporates the natural environmental factors that are unlikely to be affected by human activities, yet influence the taxa there. This model predicts the taxa that occur under least-impacted, reference conditions. The observed biotic taxa (O) at a test waterbody are compared with the taxa that the predictive model expects (E) to find in the absence of human stress, and the deviation is expressed as a ratio (O/E). Impairment is inferred if the O/E values measured at a test site fall outside the error inherent in the predicted E and estimated O/E.

In the multimetric approach, the predictive model method can provide site-specific prediction of the composition of biota in test waterbodies, the assessment requires no assumptions regarding the specific types of stress that affect biota, and it uses independent data for matching test conditions with reference conditions. The predictive modeling approach is difficult to apply in regions where not many undisturbed reference waterbodies can be found. This approach has not been tested for organisms other than benthic macroinvertebrates.

SUMMARY

Biotic indexes for evaluating the health of freshwater systems are used increasingly as they are becoming more standardized and cost-effective, and presenting their results is easier. Many human-induced changes in the physical and chemical properties of water resources are difficult to detect because of their temporal and spatial variation. Even when water resources are monitored using broad temporal and spatial coverage, physical and chemical measurements are insufficient to protect water resources because many human influences cannot be measured by using a physicochemical approach. Furthermore, biotic assemblages represent the end point of the combined influences of human disturbance on aquatic environments, and thus, biotic indexes provide a measure of overall water resource health.

Many aquatic organisms, including fish, benthic macroinvertebrates, and periphyton, can be used as indicators of aquatic ecosystem health. During the past 20 years, many IBI versions for fish, benthic macroinvertebrates, and periphyton have been developed worldwide to meet the needs of localized climate, zoogeological zone, and thermal and hydrologic regimes. All multimetric IBIs share several features. The overall index score is the sum of the scores of several individual metrics. Each metric represents a different attribute of the structure, composition, or function of the biotic

assemblage. Natural environmental or biogeographic factors that influence these attributes are taken into account in applying the index. Metrics are chosen and calibrated largely based on empirical data. Metric scores from references, those representing the least impacted waterbodies within a region, provide a standard for comparison. Each metric is sensitive to one or more types of environmental degradation. Each of the fish, benthic macroinvertebrate, and periphyton IBIs has its strengths and weaknesses. We need to use IBIs in combination with physical habitat and water chemistry assessments to establish criteria that direct human activities toward improvement and protection of water resource health.

The predictive modeling approach, such as RIVPACS, is different from the multimetric method. Instead of using multiple metrics, it uses the ratio between observed and expected for the occurring probability of aquatic organisms. However, both predictive model and multimetric approaches use regional reference sites and achieve similar goals in assessing the health of water resources.

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REVERSAL OF THE CHICAGO RIVER

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INTRODUCTION

Through the end of the nineteenth century, the rapidly growing city of Chicago was plagued with frequent epidemics of waterborne diseases. The city's death rate was one of the highest in the world. Despite the widespread installation and use of sewers beginning in 1855, the problem persisted. In fact, the sewers, while providing some adequate local drainage, only moved the public health hazard to the Chicago River and, inevitably, to Lake Michigan. As the lake also supplied the city with its drinking water, a vicious cycle existed. Successive efforts were undertaken to move the water intakes farther from shore. But, in time, the plume of pollution would reach farther out into the lake, enveloping the intakes. At this time, acceptable technology for the treatment of large quantities of potable water or sewage was nonexistent.

By the 1880s, it became clear that the only viable solution was to discharge the sewage across a subtle subcontinental divide 10 miles distant from the Lake Michigan shoreline. There, the Des Plaines River, a tributary of the Illinois and Mississippi Rivers, could be reached. A deep channel from Chicago to Joliet would be necessary. A new governmental entity, a sanitary district, was created and charged with the job of building the

channel. When its work was finished, a mere decade later, this remarkable project resulted in the excavation of 42,230,000 cubic yards of rock and soil, construction of 460,000 cubic yards of masonry for channel walls and bridge abutments, and the erection of 31 bridges. The total cost was \$33.5 million. Today, the 28-mile constructed channel is called the Chicago Sanitary and Ship Canal, although to its builders, it was referred to as the Main Channel.

COMMISSION FORMED

The City of Chicago Common Council responded to citizen demands in January 1886 by taking its first official action to solve the problem of the Chicago River. It formed the Commission on Drainage and Water Supply. The commission's charge was to outline a solution to the problem within 1 year, in time for legislative action by the Illinois General Assembly. The commission issued its 36-page preliminary report in January 1887, which presented three alternatives for sewage disposal: (1) discharge sewage into Lake Michigan away from the city, (2) dispose of sewage on land, and (3) discharge sewage into the Des Plaines River.

Cost estimates were based on a projected population of 2.5 million, roughly three times the then current population. Discharge to the lake was estimated at \$37 million, disposal on land at \$58 million, and discharge to the Des Plaines River at \$28 million. Discharge to the Des Plaines River was the recommended solution based on cost and a belief that it was a more reliable technology.

The commission set the parameters for design of a large channel from Chicago to Joliet, using as input the plans of

others as well as their own creative ideas and the results of their exhaustive investigations. Determining adequate capacity was based on meeting three fundamental needs of the city and region: storm flow, sanitation, and navigation. The commission recommended a channel cross-section area of 3600 square feet and a velocity of 3 feet per second. The resulting discharge capacity of 10,000 cubic feet per second was intended to serve a population of 2.5 million people. It was the commission's opinion that the channel capacity would prevent backflows into the lake and protect the water intake cribs located two miles offshore. The commission estimated the cost to build the channel at \$20.3 to \$24.5 million. It also recognized the potential for water power development because of the steep descent of the Des Plaines River near Lockport (see Fig. 1).

The scope of the recommendations and the size of the undertaking required the state legislature to authorize the formation of a new unit of local government. The city of Chicago was in debt to the legal limit, and the task at hand would require considerable financial resources. A new entity encompassing a larger area could borrow anew, could have a larger tax base than the city, and would have powers beyond the city limits. A legislative commission began work in May 1887, writing what would become an authorizing statute: "An act to create sanitary districts and to remove obstructions in the Des Plaines and Illinois rivers," effective July 1, 1889. The Governor of Illinois rapidly approved, opening the way for the sanitary district to be formed by referendum.

Typical of Chicago's political muscle, within 6 months, the proponents had established the boundaries of the new Sanitary District of Chicago (SDC), conducted the referendum, which passed by a whopping 70,958 to 242,

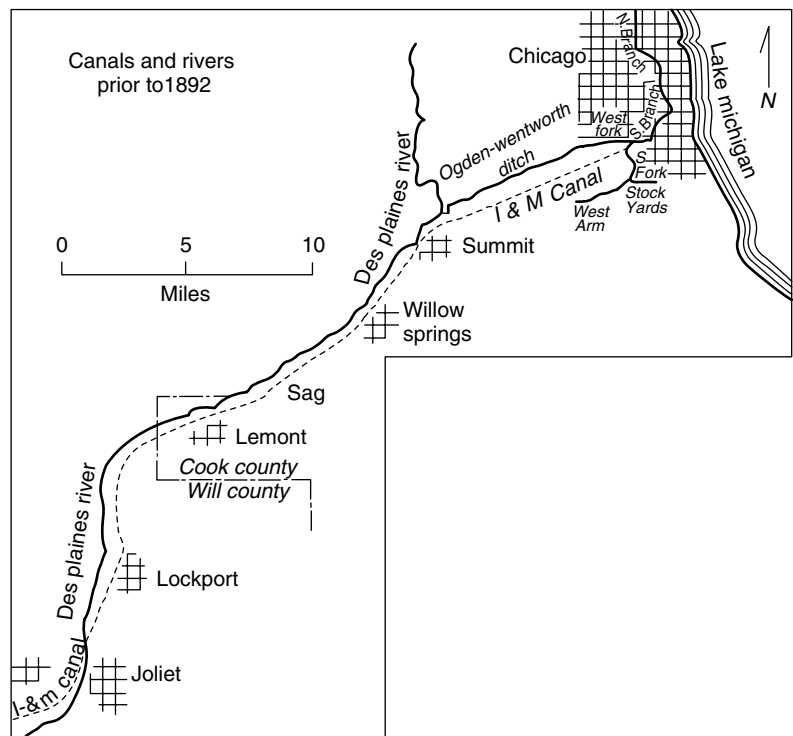


Figure 1. Canals and rivers prior to 1892.

elected nine trustees, and had them sworn in. The SDC was officially organized by January 1890. However, the new entity had to wait out two legal challenges, which went up to the Illinois Supreme Court. By June 1890, the court affirmed the authority of the SDC. Now the SDC could borrow money, approve a tax levy, and begin work.

MAIN CHANNEL WORK BEGINS

The SDC's first step was to select a Chief Engineer and direct him to present to the Board of Trustees (Board) not less than four routes of a channel to cross the 8 miles between the South Branch and Summit and to make investigations of the Des Plaines River valley. The intent was to build a channel and two pumping stations: one to lift the waters of the South Branch into the channel and the other to lift the water out of the channel into the Des Plaines River. It was also anticipated that the Des Plaines River would have to be enlarged to accept the additional flow. However, over a year went by with little progress as the board was divided and a succession of chief engineers did not survive the dissension. The third chief engineer was able to put together a plan that the board, with several new members, could agree on in January 1892.

In the spring of 1892, the board focused on getting the 14.8-mile rock section of the Main Channel designed and under construction. It was assumed that excavation in rock would be more difficult and time consuming than excavation in the earth. June 1893 saw the appointment of Isham Randolph as Chief Engineer (the fifth one), and this finally brought stability to the SDC Engineering Department as he remained in this position to 1907.

The rock section began at Willow Springs and ended at Lockport (see Fig. 2). New surveys and subsurface investigations were constantly revising knowledge of the route. The channel route was west of the I&M Canal and east of the Des Plaines River. The surface of Niagaran dolomite was very close to or formed the bed of the Des Plaines River throughout much of the reach from Sag to Lockport. From Willow Springs to Sag, the rock surface was high enough that a channel with a discharge capacity of 10,000 cubic feet per second (cfs) was required by the act. Throughout the rock section, the channel is rectangular with a width of 160 feet and a channel depth of 24 feet.



Figure 2. Excavation of the rock section.

In June, 14 contracts were awarded and work began on September 3, 1892, with a ceremony at the Cook-Will County Line. Some of the rock excavation contractors proceeded apace and their work was completed by 1895, 1 year earlier than anticipated. Other contractors were plagued with various problems, and the work was not completed until 1899.

To make room for the Main Channel between Summit and Lockport, the eastward meanders of the Des Plaines River had to be relocated; thus an aggregate of 13 miles of river diversion channels was constructed as well as a continuous levee separating the Main Channel and river. Keeping the river out of the construction area where the Main Channel was being constructed was a continuing problem for the contractors during flood periods.

Next to go under contract was the 6.2-mile earth and rock section from Summit to Willow Springs. Again, the route follows the I&M Canal and Des Plaines River. At Summit, the route of the Des Plaines River turns north, separating from the routes of the Main Channel and I&M Canal, which turn toward Chicago. Six contracts were awarded in January 1893 for channel excavation, and all work was completed by 1899. The channel was trapezoidal in cross section with a bottom width of 210 feet, side slopes of 2 horizontal to 1 vertical, and a depth of 24 feet. Because rock was encountered in this section, the act required the channel to have a discharge capacity of 10,000 cfs.

The earth section of the Main Channel extended from Summit to the West Fork, beginning just west of Robey Street (now Damen Avenue) in Chicago. As no rock was encountered in the channel cross section, the act allowed construction of a channel with a capacity of only 5,000 cfs. The act contemplated that as the population grew, the channel capacity would be increased to 10,000 cfs. This eventually happened in 1912. This lesser capacity channel had a bottom width of 110 feet, side slopes of 2 horizontal on 1 vertical, and a depth of 24 feet. This reach was notable because of the many railroad crossings. As a result of negotiations with railroads for one of the crossings, the SDC was obligated to also construct the one-third mile-long Collateral Channel connecting the Main Channel to the West Fork along the alignment of what is now Albany Avenue. The short bypass channel relieved the railroads of having two movable bridges within one-half mile of track right-of-way.

The 7.2-mile earth section reach was divided into eight contracts, the first six of which were awarded in December 1893. The last two contracts, awarded in May 1894, provided for dredging and transporting the spoil by scow to the lakefront to be used as fill for the creation of what is now Grant Park. Other spoil was deposited in what is now Douglas Park and used as fill for local streets and boulevards. Because of the delayed construction of a major railroad crossing, dredged channels extended from the West Fork to Western Avenue along the route of the Main Channel and to the Main Channel north embankment along the Collateral Channel. Work on these eight contracts was completed in either 1898 or 1899.

To control the discharge of water from the Main Channel, two more contracts were awarded in 1895 and 1896 for construction of the Lockport Controlling Works.

These works consisted of seven vertical gates, each 30 feet wide and 20 feet high, and a 160-foot long sector-type dam, called the Bear Trap Dam. The dam could be lowered to allow flow over its top, providing for sensitive discharge control. The gates were raised to provide for rapid increases in discharge over a short time period. These two types of control were necessary because the end of the Main Channel was 35 miles from Lake Michigan and the lake level could change rapidly on short notice. These control structures were among the largest in the world at the time, comparable with controls on the outlet of Lake Superior and on the Ohio River.

THE JOLIET PROJECT

Near Lockport, the Des Plaines River began a relatively steep descent to Joliet, at some places flowing over exposed dolomite. Once in Joliet, the river was joined by the I&M Canal where several successive dams created navigation or power pools. Planning for a channel through Joliet would require more study and dealing with the I&M Canal Commissioners and the City of Joliet. Complicating the matter was the physical setting north of Joliet. The river channel meandered from east to west across the valley floor; the I&M Canal and two railroads were on the east side of the valley, and several industries with water power developments were along the river.

The confluence of the new channel and the I&M Canal required modifications to the tow path, locks, and dams, but the canal commissioners were slow to come to terms with the SDC. The SDC proceeded with work and was sued by the canal commissioners. Three contracts were awarded in early 1898 for the 5.1 miles of work. Because of the lawsuit and other contract problems, the work was not completed until 1901. However, enlargement of the Des Plaines River channel capacity was completed by late 1899.

SOUTH BRANCH IMPROVEMENTS

By August 1895, 28 miles of the Main Channel were under contract, and rock excavation was nearly complete near Lemont. The SDC needed to improve the capacity of the Chicago River and the South Branch so that it could deliver the flow of water from Lake Michigan as required by the act. The river had many bends and constrictive bridge openings, was shallow in spots, and was always busy with boat traffic. Fortunately, no time would be consumed in debate over the route.

The U.S. Army Corps of Engineers (Corps) had plans to improve the Chicago River, so the SDC concerned itself with improvement of the 5-mile reach of the South Branch between Lake and Robey streets. All work in the South Branch was subject to permits issued by the Corps, but the SDC would often begin work before a permit was issued. One contract was awarded in May 1897 for removal and replacement of dock walls and dredging to pass 5000 cfs in a channel 200 feet wide and 20 feet deep. Other contracts were awarded in 1898 for replacement of two center pier bridges with restrictive openings and a large

bypass conduit around a restrictive bridge opening that could not be enlarged. All work was sufficiently completed by 1899 to provide for the design capacity.

The act made reference in many locations to a navigable waterway, maximum velocities, and minimum depths and widths, all of which defined the conditions for safe commercial navigation. However, the act made no reference to bridges. Lacking statutory definition, the matter of bridges was discretionary to the SDC, which caused much debate and division among the members of the board. As there was no navigation lock at Lockport, navigation on the Main Channel was not an immediate priority. The bridges were put under contract late in the 1890s, and all substructure work was completed by late 1899 before the Main Channel was placed in service in January 1900. The SDC built or funded the construction of 31 bridges to effect the reversal of the Chicago River. The 13 bridges over the Main Channel were designed and constructed to eventually be made movable to allow for passage of boats.

REVERSING THE FLOW

To place the empty 28-mile-long Main Channel into operation, water would need to be added slowly so as not to cause damage by rapidly rising water levels or swift currents. Despite not having specific approval from the Governor, water was let in at the Chicago end beginning on January 2, 1900, through a wooden flume in the earth dike across the south end of the Collateral Channel. The filling continued to January 14 when the water level in the Main Channel reached the water level in the West Fork. The next day, the earth dam across the Main Channel west of Western Avenue was cleared away by dredges and the waters on each side came together. After 13 days of filling, the water level came to rest, to wait for the Governor's approval to discharge at Lockport.

A special commission had been appointed by the Governor to inspect the work and advise on satisfactory completion of the work. The SDC vowed to wait for the Governor's approval before releasing water from the Main Channel at Lockport. The Board and commission members traveled to Lockport on January 17, 1900, to be at the Lockport Controlling Works when the approval came. The Governor's approval was received by telegram, and the Bear Trap Dam was lowered slightly below the water level to allow a thin sheet of water to flow over its top (see Fig. 3). After a brief ceremony, the valves controlling the dam were opened and the massive 160-foot-long dam disappeared beneath the water. A torrent of water rushed out of the Main Channel over the dam toward the Des Plaines River. On this chilly day in the first month of the new century, slightly more than 10 years after passage of the authorizing act and after more than 7 years of construction, the Main Channel was now in operation to save Chicago from its own waste.

The reversal of the Chicago River has been supplemented with other works, and the diversion of water from out of the Great Lakes basin has been the subject of extensive litigation, but the reversal of the river has never been interrupted.



Figure 3. A total of 4250 cubic feet per second passes over the Bear Trap Dam.

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FLOOD CONTROL IN THE YELLOW RIVER BASIN IN CHINA

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GENERAL INFORMATION ON THE YELLOW RIVER

The Yellow River (see Fig. 1), the second longest river in China originates in the Yuguzonglie Basin in the Tibet Plateau in west China and enters the Bohai Sea in the east. It is 5464 km long, has a water level difference of 4480 m, and the basin area is 795,000 km², 8.28% of China. In the entire Yellow River Basin, 98 million people are settled.

The most important water resource in the Yellow River Basin is precipitation. The annual precipitation is concentrated in the period from July until October. The average annual runoff of the Yellow River is 58 billion m³ (1987). About 10 to 20% of the runoff is discharged from March until June, and 60% is discharged in the flood season from July until October. The interannual runoff distribution is also not uniform. The ratio between the maximum and the minimum value can be up to 3.4. Since 1919, a dry period that was longer than 5 years occurred



Figure 1. Location of the Yellow River Basin in China.

twice. The annual average runoff during these dry periods was 39 billion m³. The total amount of groundwater in the basin is 40 billion m³ (excluding the interior basin). The exploitable amount of groundwater is 12 billion m³. The distribution of groundwater is not uniform in the basin (1).

GENERAL CONDITIONS FOR FLOODING AND SEDIMENTATION

The frequent occurrence of extensive flood disasters is related to its geography (high in the west and low in the east) and to its uneven temporal and spatial distribution of precipitation. As recorded since the third millennium B.C., the Yellow River has overflowed its dikes 1590 times and changed its course 26 times. Nine of those were major course changes. The range of the course divagation covers an area of 250,000 km². Thousands of people were killed in many floods. The cycle of flooding and death earned the Yellow River the name "China's Sorrow."

Flooding in the Yellow River Basin

There are two kind of floods in the Yellow River: storm-caused floods that occur between July and October and ice-jam floods that normally occur in February (1). The main floods of the Yellow River occur in July and August. These are peak floods of short duration that rise and fall quickly. Historical investigations show that the largest flood occurred in 1843; the peak discharge was 36,000 m³/s. The maximum flood measured occurred in 1958; the peak discharge was 22,300 m³/s at Huayuankou; see Fig. 2 (1). This flood is used to determine the design level of the lower reach. The design level at Huayuankou hydrologic station is 22,000 m³/s, and the probability is once in 1250 years. The construction of the Xiaolangdi dam is not included in



Figure 2. Main hydrology station in the Yellow River Basin (Source: Yellow River Water Conservancy Commission).

this design level. Table 1 shows the total flood frequency analysis for the lower reach of the Yellow River.

In spring, ice-jam floods often occur between Huayuankou and the estuary. The riverbed slope is shallow in this part, so the velocity of the water is low. The flow direction from west to east changes from a low latitude to a higher latitude at the Bohai Sea. This results in a higher temperature at Huayuankou than in the estuary. The average winter temperature near the estuary is 3.4°C lower than at Huayuankou. For this reason, in 80% of the years, the river freezes, and because of this, in most of these years, ice-jam floods happen. According to the statistics, between 1950 and 1983, there were 29 years in which the river froze. In 1951 and 1955, serious ice floods occurred. The length of the frozen river was 550 km. and 623 km. respectively. In history, the ice-jam floods frequently broke dikes. According to rough statistics between 1883 and 1936, ice-jam floods broke dikes in 21 years.

The extreme floods measured at Huayuankou are from different geographical sources, and these floods also have different characteristics. A classification has been made for the different floods based on historical floods and for a few floods shown in Table 2. The classification is as follows:

- The inflow upstream of the Sanmenxia reservoir and comparatively small inflow downstream of the Sanmenxia reservoir mainly compose the flood. This type of flood has a high flood peak, a large flood discharge, and high sediment concentration.

- The flood is composed mainly of the inflow downstream of the Sanmenxia reservoir. This type of flood has a rapid flood rise, a high flood peak, low sediment concentration, and a short forecast period.
- Half the flood is composed of the inflow upstream of the Sanmenxia reservoir and half of the inflow downstream of the Sanmenxia reservoir. This type of flood has a small flood peak, a long duration, and low sediment concentration.

Sediment in the Yellow River Basin

The majority of the sediment sources in the basin are situated in the upper and the middle reaches. In the upper reach, the average sediment concentration is only 6 kg/m^3 , and the average sediment transport is 142 million tons. In the middle reach, the river flows through a loess plateau. Due to the fine grain sizes and the restricted vegetation in the loess plateau, soil erosion can cause a huge sediment load, especially during storms. During a storm, the sediment concentration in the river can rise to more than 500 kg/m^3 (1), as illustrated in Fig. 3.

One of the important characteristics of the Yellow River is that water and sediment originate in different regions. In the upper reach, the annual sediment input is 8.7%, whereas the annual runoff is 54% of the annual input. In the middle reach, the annual runoff is 36%, whereas the sediment input is 89%. From Xiaolangdi to the Bohai Sea, the lower reach, the surface area is limited, and the annual runoff depends mostly on the tributaries, the

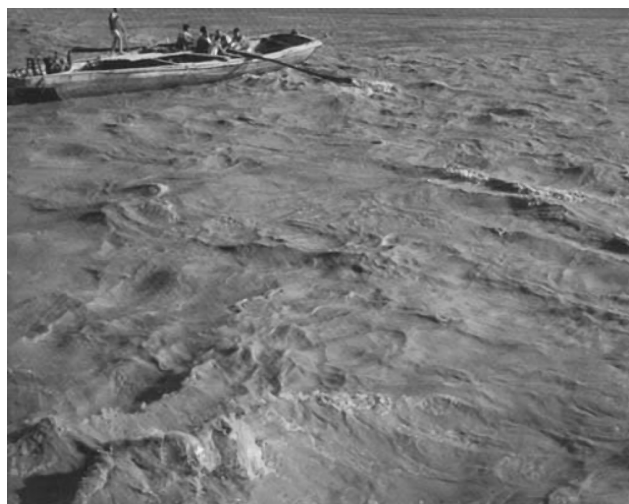
Table 1. Flood Frequency Analysis at Huayuankou^a

Item	Unit	Average	$p = .01\%$	$P = 0.1\%$	$P = 1.0\%$
Flood peak	m^3/s	9780	55000	42300	29200
5d flood discharge	b m^3	26.5	12.5	9.84	7.13
12d flood discharge	b m^3	53.5	20.1	16.4	12.5
45d flood discharge	b m^3	153	41.7	35.8	29.4

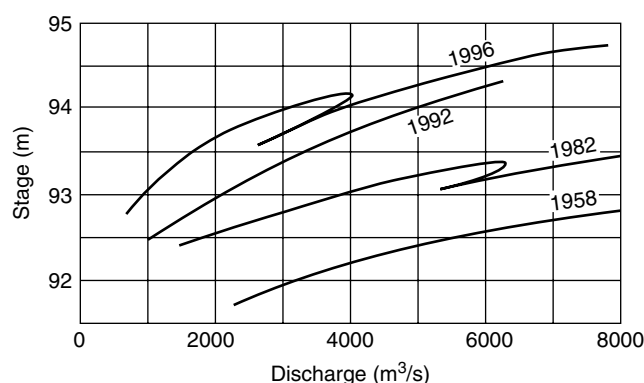
^aReference 1.

Table 2. Composition of the Floods in Huayuankou^a

	Year	Huayuankou		Sanmenxia		
		Discharge, m ³ /s	12-d Discharge, billion m ³	Discharge, m ³ /s	Huayuankou, m ³ /s	12-d Discharge, billion m ³
Upstream of Sanmenxia	1843	33,000	13.6	36,000	30,800	11.0
	1933	20,400	10.1	22,000	18,500	9.18
Downstream of Sanmenxia	1761	32,000	12.0		6000	5.0
	1958	22,300	8.9		6400	5.15
Up- and down stream of Sanmenxia	1957	13,000	6.6		5700	43.1

^aReference 1.**Figure 3.** Yellow River: High sediment concentration flow (2).

Yilouhe and the Qinhe. The annual runoff of the lower reach is 10%, and the sediment input is 2% of the annual input. Due to severe soil loss in the middle reach, on average, 1.6 billion ton of sediment enters the river channel at Huayuankou annually, of which about 1.2 billion ton is carried to the estuary region, leaving behind a substantial amount to contribute to the silting of the river channel (1). This results in high deposition in the riverbed of the lower reach. Due to this, the riverbed fills up, on average, 0.1 meter every year. By the time, the Yellow River reaches Kaifeng, it runs 10 m above the surrounding plain. Some Ming and Song dynasties' artifacts 500 to 800 years old lie under 5 to 8 m of silt at Kaifeng City. The present riverbed above its floodplain is about the same as the old course that was breached and abandoned in 1855 (<http://www-geology.ucdavis.edu/GEL115/115CHXXYellow.html>). Therefore, nowadays, the Yellow River is called a "suspended" river. In Fig. 4, the Q/h relation for different years at Huayuankou is shown. This figure shows the impact of the decreasing flood conveyance due to the rise of the riverbed. Even for a normal peak flood of 5000 m³/s the water level rose about 2 meters in the period 1958–1996. It is expected that the process of bed level raising will continue in the near future.

**Figure 4.** Water level-discharge relation for three floods (3).

FLOOD CONTROL IN THE YR BASIN

Historical Flood Management in the Yellow River

As the most sediment-laden river of the world, the extreme sediment load of the river causes problems in water resources management and flood protection (1). The first inhabitants of the Yellow River floodplains and the surrounding areas lived with floods. They escaped from flood inundation by living in highlands or moving far away from flood-prone areas. Legends say that before the first manager of Chinese waters, Yu The Great (twenty-first century B.C.), built dikes on the lower reaches, barriers to block or store the water in large holding areas were tried for a long time. Yu The Great was great because of his innovation to direct flow to the sea by dikes. The first recorded river training by construction of levees started probably in the Warring State periods (770–221 B.C.) when the Yellow River was not yet a suspended river.

During the Han Dynasty (206 B.C. –220 A.D.), disasters of flood breaches and course shifting were more overwhelming than ever before along with sediment deposition in the lower reach as well as more intensive human activities in the loess plateau and in the floodplain. And the Yellow River lower reach has gradually become a suspended river. Because the heavy silt load cut the effectiveness of flood control, it is of key importance that enormous constant efforts are made to maintain equilibrium with the river. To tackle the flooding in the Yellow River, one prominent water manager Ja Rang stated his "three measures on

River treatment (flood management)” around 7 B.C. The first measure, the best option according to Jia Rang, was to return the river to the abandoned course. However, this idea proved to be impossible at that time because of the limited technological capability when the abandoned channel was seriously silted. The second measure was to dissipate the power of the river by draining off water for irrigation in the lower reach and diverting relatively silt-free streams into the Yellow River to increase the silt-carrying capacity. The last measure was to strengthen dikes to contain the river. This measure was his last choice. He believed that the endless need for dike raising was a waste of time, labor, and resources.

After Ja Rang, Wang Jing proposed and carried out river channel stabilizations in 69 A.D. so effectively that there were no major dike breaches for the next nearly 1000 years. His methods included dredging, strengthening the levees at dangerous points, digging new channels for tributaries in rough terrain, and building numerous sluice gates. A thousand years later, Pan Jixun (1521–1595) was remarkable in the Ming Dynasty (1368–1644) because he advocated building strong close dikes to contain the river so that it would scour its own narrow channel. This was the first forceful statement against the ancient principle of dividing the flow to dissipate the river’s power. This strategy was also followed by Jin Fu in the early Qing Dynasty (1644–1922 A.D.) to stabilize the river for a certain period.

Present Flood Control in the YR

During the early decades of the twentieth century, flood control in the Yellow River basin has evolved as a combination of traditional management practices, influences from European-based industrial society, and new technical and organizational features from modern Chinese society. The views on flood management were determined by the famous Chinese flood managers, Li Yizhi, Shen Yi, Zhang Hanying, and Wang Huayun. In the mid-1950s, a multipurpose plan for permanent control of the river was initiated. This plan included the construction

of more than 40 dams and projects to moderate the river’s flow (and produce energy).

Today, on the upper and middle stream of the Yellow River, there are 173 large and medium-sized reservoirs whose total storage capacity is 55.2 billion m^3 . Downstream of the Sanmenxia reservoir (see Figs. 2 and 5), two large flood retardation basins—the Beijinti and the Dongpinghu—were constructed. Their storage capacity is 2 billion m^3 each and detention area 2316 km^2 and 627 km^2 , respectively. Altogether 5000 bank protection works with a total length of 585 km were built. In this way, the flood control works can resist peak discharges of 22,000 m^3/s (1958 flood) at the Huayuankou hydrologic station (1958 flood), corresponding to a 60-year return period. The Xiaolangdi Reservoir whose a storage capacity is 12.65 billion m^3 on the lower reach increased the protection standard there to a return period of 1000 years (4).

The flood control engineering system that has been gradually formed since 1950 is to retain water in the upper and middle reaches, drain water at the lower reach, and divert and detain water on both sides of the river, guided by the notion of “stabilizing the flow by widening the channel.” The lower reach flood control system is shown in Fig. 5. The engineering works for retaining water include

- the Sanmenxia reservoir at the main course of the Yellow River, which controls 91.4% of the total basin area;
- the Luhun reservoir and the Guxian reservoir on the Yi River and Luo River, designed to reduce the flood risk for the lower reach of the Yellow River and the city of Luoyang, combined with the Sanmenxia and Xiaolangdi reservoirs; and
- the Xiaolangdi reservoir which controls 92% of the total basin area and has a total capacity of 12.65 billion m^3 and a long-term effective capacity of 5.1 billion m^3 .

Currently, both structural and nonstructural measures are adopted in Yellow River flood control. Structural measures include reservoirs, diversion structures, retardation

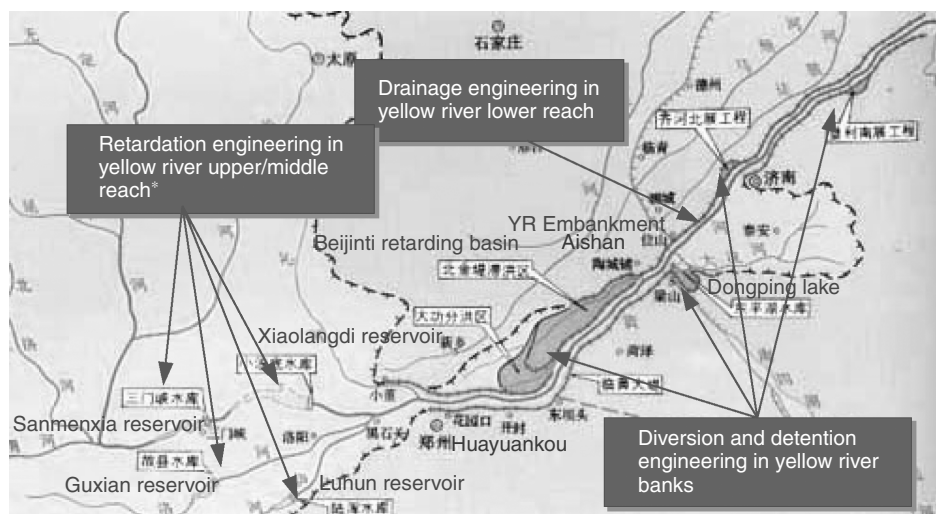


Figure 5. The flood control system of the Yellow River lower reach. Note: The upper reach’s retardation engineering is not shown in this figure.

basins, embankment, dredging, and channel modifications. Nonstructural measures include the flood control organization system, the flood control communication system, the hydrology monitoring and forecasting system, the flood regulation, command and decision making support system, the management of flood plain, and the diverting and detaining area.

Measures for Reducing Flood Risk

As stated above, the Yellow River delivers about 1.6 billion tons of sediment per year to the lower reach, resulting in an average content of 35 kg/m^3 . In the reaches with the highest concentration rates, the river contains more than 10 million kg of sediment per kilometer of which 90% is from the loess plateau in the middle reach of the river between Longmen and Tongguan. Due to the heavy sedimentation, the bed level rises in the lower reach. Where in the past the river changed its course when the riverbed was too high for the surrounding area, at present embankments, buttresses, and control works harness the river and fix the river in its place. The sediment has no place to go but deposit in the lower reach. Due to human activities such as deforestation in the middle reach, soil erosion enhances the sedimentation process in the lower reach even more. At present, the riverbed is as high as 5 to 10 meters above ground level, creating a situation where in case of a dike breach during a flood, the damage can be enormous. Reducing this flood risk can be done in two ways, by reducing the peak flow and by increasing the channel capacity.

Reducing peak flow can be and presently is achieved by retention of water upstream, by detention in basins in the lower reach and by controlled flooding on floodplains. The problem of retention is difficult coordination with other water users, among whom are the Ministry of Power and farmers; when water is to be released to create a buffer and when water is to be retained to relieve downstream flooding, to increase the power generating capacity, and to conserve water for demand in the dry season.

Capacity can be increased by reducing sedimentation and by raising the embankments. Increasing capacity is currently attempted by four types of measures:

- dredging to enlarge flow cross section.
- decreasing erosion by soil conservation projects. Large afforestation projects in the middle reach have started to conserve the soil from eroding. The main focus is on simultaneous development of agriculture, forest, and animal husbandry.
- prevention of transport of sediment to the lower reach by trapping sediment in large basins or reservoirs. The recently built Xiaolangdi reservoir is constructed to trap sediment for 10 to 15 years, until the reservoir is as small as 30% of its original size. This will prevent about eight million cubic meters of sediment from flowing into the lower reach.
- increase capacity by raising the embankments, but there are financial as well as physical limits to keep doing so.

Despite all these measures, the problems remain. Even though the flood risk is momentarily decreased by Xiaolangdi, in 15 years, the risk will reach the same level again and will increase every year. Besides this, flooding in an area of $120,000 \text{ km}^2$ between Xiaolangdi and Huayuankou is still uncontrolled. A maximum of $13,000 \text{ m}^3/\text{s}$ can be transported in this section, which corresponds to a 1 in 100 year event.

Analysis of Flooding in the Lower Reach of the Yellow River

Looking at the water resources system of the lower reach of the Yellow River, as described in the last sections, it can be concluded that 50 years have passed now without any dike-outbreak flooding in the lower reach. Two of the reasons are the three dike elevations on the lower reach and relatively low discharges. It cannot be concluded, however, that the danger of dike-outbreak flooding has been sufficiently prevented. In fact, the risk is still there, and a number of factors have even increased the risks. The following are reasons that the risk is still there (5):

1. Low design flood discharge. The present design flood discharge of the Yellow River on its lower reach is $22,000 \text{ m}^3/\text{s}$ at Zhengzhou with a probability of 2.2% (without taking into account the Xiaolangdi reservoir), taking all the dikes and detention basins into account.
2. The construction of the Xiaolangdi reservoir. If flood control and silt reduction become the most important functions, the Xiaolangdi project can improve the situation for about 15 years. Around 2015, new measures have to be adopted to keep flood control in the lower reach at least at the same level.
3. Poor quality of flood control works. There are 832 km of dike sections whose height is lower than the design water level on account of the steady filling up of the riverbed. There are 340 km of weak dike sections.
4. The strongly varying width of the river between Xiaolangdi and Gaocun. The width varies between 5 and 20 km.
5. Increasing danger of earthquakes. The lower reach of the Yellow River is located in a strong faulting area. It is predicted that in a large part of the area an earthquake of strength 7–8 on the Richter scale may occur within the next 50 years, and there is even a possibility of earthquakes of strength 9 in some areas (6).

These are the reasons that the risk has increased and is still increasing:

1. Densely populated detention areas, whose population is still growing. Beijindi detention area and Dongping Lake house millions of people and are therefore not suitable any longer to store water in case of a flood with a short forecast period (less than 24 hours).
2. Inside the flood-prone areas, approximately 78 million people live. The industrial development is high in these areas, and the economic importance

of these areas is high. The potential for flood damage has increased enormously in recent years, so a dike breach would cause a disaster.

3. The increasing number of people who live in the floodplain. The total number of people who live outside the dikes in the lower reach is 1.8 million. In Henan province only, it is already 1.10 million. These people are very seriously threatened by floods.
4. Decreasing flood conveyance capacity of the river channel. Owing to the construction of reservoirs on the upper reach and irrigation development in the whole basin, discharge in the main channel of the Yellow River has decreased and sedimentation on its lower reach has increased considerably in recent years. As a result, the water level for the same discharge is much higher than before.
5. The role of the YRCC in the distribution decisions for the reservoirs. The Sanmenxia reservoir, for instance, is under the jurisdiction of the YRCC. The management of Xiaolangdi project is not yet known. If it is run under one set of operating priorities, it can improve the flood control in the lower reach. On the other hand, a different set of priorities may enable Xiaolangdi to deliver economically interesting hydropower.

Overall, it can be said that due to the decreasing flood conveyance and the low design flood discharge, the probability of a serious flood increases rapidly. If a flood occurs, the structural flood prevention measures such as dikes and detention areas do not meet the required standard or cannot be used. The increasing population and the economic development would cause much more economic damage in case of flooding than 10 years ago. Moreover, it can also be expected that the number of casualties will be much higher than 10 years ago. The laws and the executive authorities are not able to control the number of people living outside the dikes and related unspecified dike building inside the flood-plain. So still a lot has to be done to improve flood control in the lower reach.

Looking at the measures taken to control the river in the last 50 years, the approach focuses on flood prevention by raising and strengthening the embankments and the construction of dams like Sanmenxia and more recently Xiaolangdi. However, it seems that the traditional methods cannot find sustainable solutions and create a sustainable and safe WRS for the LYR and the flood-endangered areas around this part of the river. The danger of floods is still there and, for the reasons mentioned before, the danger is even increasing.

For the future, different research projects have started, and measures are proposed. Bypasses are also suggested. Theoretically, one could dig out an alternative route for the river, complete with levees. Some scholar even suggested where the bypass can be located and which region would benefit from the Yellow River discharge from a regional economic and ecological view point. However, such an artificial bypass would be incredibly expensive, and the consequences for social activities and ecosystems are uncertain.

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CHIRONOMIDS IN SEDIMENT TOXICITY TESTING

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INTRODUCTION

Historically, assessments of water quality have largely been based on the evaluation of water-borne contaminants and water quality criteria established by comparing aqueous concentrations of compounds to their toxicity determined in water-only exposures. It was not until the mid-1970s that it was realized that contaminants, particularly hydrophobic compounds, also occurred in sediments, often at high concentrations. With this new understanding came the realization that water quality criteria based solely on aqueous concentrations were insufficient to regulate contaminants that existed predominantly in sediments and would, therefore, not adequately protect benthic organisms. To develop appropriate and effective criteria for regulating sediment-associated contaminants, the need to develop new methods capable of assessing sediment toxicity became apparent. This need led to the birth of the field of sediment toxicology.

Since it was first recognized that sediments served as both a source and a sink for many contaminants, research in the field of sediment toxicology has largely focused on the development and validation of methods to assess toxicity. Detailed reviews that document the collection and manipulation of sediments, selection of test species, development of test methods, optimization of test conditions and experimental design, and validation of freshwater

sediment toxicity test methods are provided in Nebeker et al. (1), Giesy and Hoke (2), Burton (3,4), Ankley et al. (5,6), Ingersoll et al. (7,8), and Diamond et al. (9). A critical consideration in the development of sediment toxicity tests is the selection of appropriate test species. Although several invertebrate species now play an integral role in the assessment of sediment toxicity in freshwater systems (10), arguably none have been used more extensively than the two midge species *Chironomus tentans* and *C. riparius*.

The genus *Chironomus* (Diptera: Chironomidae) includes a large number of species globally, many of which occur ubiquitously in aquatic habitats. To date, seven species of *Chironomus* have been used in sediment toxicity testing: *C. tentans* (nearctic populations of *C. tentans* are now known as *C. dilutus* [Shabanov et al. (11)], *C. riparius*, *C. plumosus*, *C. attenuatus*, *C. prasinus*, *C. crassiforceps*, and *C. tepperi*. Of these, *C. tentans* and *C. riparius* have dominated the sediment toxicology literature, with the former being used predominantly in North America and the latter predominantly in Canada and Europe. Both species have been incorporated into standard test methods (12–14), and their application in sediment toxicity testing has been reviewed in detail (10,12,15). The adoption of these two midges in sediment toxicity assessments reflects their importance to the food webs of aquatic systems (e.g., ecological relevance), relative sensitivity to contaminants, tolerance of a wide range of sediment characteristics, and benthic mode of existence (8). From a practical standpoint, both midges have short generation times and are therefore easy to culture and available for year-round testing, are relatively insensitive to manipulation during culturing and toxicity testing, and have wide acceptance among regulatory agencies.

In light of the dominant application of both *C. tentans* and *C. riparius* in sediment toxicity testing, including the availability of standard protocols, this review will focus on these two species, with the goal of providing a pragmatic overview of their application in sediment toxicity assessment. For information on the application of the other species of *Chironomus* listed above in toxicity testing, the reader is referred to Sanchez and Tarazona (16) (*C. prasinus*), Wilson et al. (17) (*C. tepperi*), Peck et al. (18) (*C. crassiforceps*), Fargasova (19) (*C. plumosus*), and Darville and Wilhm (20) (*C. attenuatus*).

BIOLOGY OF *C. TENTANS* AND *C. RIPARIUS*

The biology of *C. tentans* and *C. riparius* has been described in detail elsewhere and is thus only briefly described here. For information of the biology of *C. tentans*, the reader is referred to Sadler (21), Hall et al. (22), Ineichen et al. (23,24), Sibley et al. (25,26), and Watts and Pascoe (27). For information on the biology of *C. riparius*, the reader is referred to Edgar and Meadows (28), Caspary and Downe (29), Downe (30), Rasmussen (31,32), Watts and Pascoe (27), and Hooper et al. (33).

C. tentans and *C. riparius* are holometabolous insects and thus have a life cycle that consists of an egg, larval, pupal, and adult stage. The larval stage in *C. tentans* lasts approximately 23 days at the recommended test

temperature of 23 °C (34). In contrast, the larval stage of *C. riparius* lasts between 15 and 18 days at a temperature of 20 °C, depending on the life stage (e.g., egg versus newly hatched larvae) that initiates the test (35). However, in a study comparing the relative sensitivity of the two midges to selected contaminants, Watts and Pascoe (27) found that the duration of the larval stage for *C. riparius* and *C. tentans* was comparable at 22 °C. Larvae of both species pass through four instars and are generally comparable in size (based on head capsule width) up to the fourth instar. Thereafter, larvae of *C. tentans* grow considerably larger than those of *C. riparius* (27). Toward the end of the fourth instar, larvae of both species become inactive and cease to grow; this period is followed by pupation, which lasts 1–2 days. In both species, emergence follows a bimodal pattern (protandry) in which males emerge 2–5 days before females (26,35). Males can be distinguished from females by the presence of plumose antennae. Males of both *C. tentans* and *C. riparius* are capable of multiple matings, whereas females are not receptive to additional matings once they have been inseminated (26,30). Impregnated females of both species oviposit a single egg mass the day after insemination. Female *C. tentans* will occasionally produce a small secondary egg mass, but these eggs are usually not viable. Although *C. tentans* has been reported to produce egg masses containing up to 2300 eggs (22), 800–1000 is more typical (25,26). In contrast, egg masses of *C. riparius* typically contain between 400 and 600 eggs (36). Unmated females generally do not oviposit and resorb the egg mass. However, unmated females of *C. tentans* may occasionally produce an egg mass containing nonviable eggs (25). In *C. tentans*, adult females live up to 5 days, whereas males live up to 7 days.

ACUTE SEDIMENT TOXICITY TESTS

The most commonly applied standard test using both *C. tentans* and *C. riparius* is the 10-d survival and growth assay (1,7,10). The 10-d version using *C. tentans*, along with its 10-d counterpart using the amphipod *Hyallela azteca*, has been evaluated in an interlaboratory study of precision (3). The procedures and schedule for conducting a 10-d test are provided in Table 1. Although the test procedures described below and in Table 1 are those for *C. tentans*, the test conditions and requirements are similar for both species. The reader is referred to ASTM (13) and Environment Canada (14) for specific test methods for *C. riparius*.

The 10-d test method for *C. tentans* consists of exposing 10-d old larvae to a contaminated sediment at 23 °C with a light:dark photoperiod of 16:8 and an illumination of 500–1000 lux (7,12). Test chambers consist of 300 ml beakers containing 100 ml of sediment and 175 ml of overlying water. The number of replicates recommended for an assay is eight per treatment, although fewer (not less than 3) may be used. Each beaker is fed 1.5 ml of a 4 g/L Tetrafin fish food slurry each day. After the 10-d exposure period, larvae are sieved from the sediment and enumerated to determine survival. A test is considered to be acceptable if survival in the control or reference sediment is $\geq 70\%$ (Table 2). If growth is to be measured,

Table 1. Test Setup and Activity Schedule of the *C. tentans* Acute Sediment Toxicity Test

Test Day	Activity
-1	Add 100 ml of homogenized sediment (or fine sand if water only exposure) to each replicate beaker and place in appropriate holding tank. Allow sediment to settle (about 1 hr), and then add 1.5 ml of Tetrafin fish food slurry (4 g/L) to each beaker.
0	Randomly allocate 10 larvae (50% must be in third instar) to each replicate beaker using a Pasteur pipette. Let beakers sit for 1 hr after addition of the larvae. After this period, gently immerse/place all beakers into the exposure system.
1-end	On a daily basis, check assay system and add 1.5 ml of food to each beaker. Dissolved oxygen should not fall below 2.5 mg/L. Reduce feeding rate or frequency if persistent low-dissolved oxygen occurs.
10	Sieve the sediment from each beaker to recover larvae for growth and survival determinations. Place recovered larvae from each replicate in preashed aluminum pans and oven dry for 24 hours at 90 °C.
11	Weigh dried larvae to the nearest hundredth milligram. If ash-free dry weight is to be determined, ash the pan and larvae at 550 °C for 2 hours. Allow pans to cool to room temperature in a desiccator and reweigh. Individual larval dry weight is determined as the difference between the total mean dry weight and total mean ashed weight divided by the number of surviving larvae per replicate.

Adapted from Ingersoll et al. (7).

larvae are placed in pre-ashed aluminum weigh pans, dried at 90 °C for 24 hours, and weighed once they have returned to room temperature. In instances where dry weight may be influenced by different sediment particle size characteristics, larval weight should be determined as ash-free dry weight (37). In this approach, the dried larvae in the aluminum pan are ashed at 550 °C for 2 hours and reweighed as above. The ash-free dry weight is determined as the difference between the dried weight and ashed weight. Larval weight is typically expressed on an individual basis (e.g., milligram dry weight per individual). However, Call et al. (38) showed that dry weight varied depending on the number of surviving larvae, with higher weights recorded in sediments with lower survival. They found that this bias was removed by expressing larval dry weights as total weight per replicate.

Once collected the data must be analyzed statistically. A detailed account of statistical procedures for analyzing data generated from both acute and chronic (see below) toxicity tests is beyond the scope of this document. For general guidance, the reader is referred to U.S. EPA (12) and Environment Canada (39).

Based on a comparison of LC50s, Watts and Pascoe (27) compared the relative performance and sensitivity of *C. tentans* and *C. riparius* in sediment toxicity tests with cadmium and lindane. They found no difference in sensitivity in exposures to cadmium but showed that *C. tentans* was significantly more sensitive than *C. riparius* to lindane. The latter result is consistent with the findings of Pauwels and Sibley (10) who compared the relative sensitivity of several sediment test species based on LC50 values obtained from a survey of the sediment toxicology literature. In most cases, *C. riparius* was found to be less sensitive than *C. tentans*.

CHRONIC SEDIMENT TOXICITY AND LIFE CYCLE TESTS

In response to the need for chronic evaluation of sediment toxicity, Benoit et al. (34) introduced a life cycle test using *C. tentans*. A parallel test was developed using the amphipod *Hyallela azteca* (40), and collectively, these two assays represent the only standardized tests currently available for evaluating the effects of sediment-associated contaminants over the full life cycle of a benthic organism (12,13). Although a long-term toxicity test using *C. riparius* has apparently been developed (41), this test has not been standardized and appears to have received little attention in recent years. Nonetheless, *C. riparius* has been applied in several long-term studies that have incorporated emergence and reproduction (42,43). Given its close similarity to *C. tentans* in most aspects of its life history, *C. riparius* could probably be readily adapted for use in life cycle testing following the standard procedure for *C. tentans*.

The recommended exposure system, required testing apparatus, and experimental design for conducting a life cycle test with *C. tentans* is described in detail in Benoit et al. (34) and U.S. EPA (12). Table 3 summarizes the activity schedule for a typical life cycle test. The test is initiated with first instar larvae that are <24 hours old. The egg cases are transferred to a glass petri dish (do not use plastic petri dishes as the larvae will quickly

Table 2. Test Acceptability Criteria (Survival and Growth) and Recommended Criteria (Emergence, Reproduction, and Percent Hatch) for Control Organisms in 10-d Tests with *C. riparius* and *C. tentans* and the *C. tentans* Life Cycle Test

Endpoint/Condition	10-d Test		Life Cycle Test
	<i>C. tentans</i>	<i>C. riparius</i>	
Survival	70%	70%	70% ^a
Growth	0.48 mg afdw ^b /individual	Not determined	0.60 mg dry weight/individual or 0.48 mg afdw/individual
Emergence	N/A	N/A	≥50%
No. of eggs/egg mass	N/A	N/A	≥800
Percent hatch	N/A	N/A	≥80%

Adapted from USEPA (12) and ASTM (13). N/A = Not applicable

^aMeasured at 20 days.

^bafdw = ash free dry weight.

Table 3. Test Setup and Activity Schedule for Conducting a Life Cycle Chronic Sediment Toxicity Test with *C. tentans*

Test Day	Activity
-4	Start reproduction flask with cultured adults (10:30 male:female ratio).
-3	Collect egg cases (6–8) and incubate at 3 °C.
-2	Check egg cases for viability and development.
-1	Check egg cases for hatch and development. Add 100 ml of homogenized sediment (or fine sand if water only exposure) to each replicate beaker, and place in appropriate holding tank. Allow sediment to settle (about 1 hr), and then add 1 ml of Tetrafin fish food slurry (4 g/L) to each beaker.
0	Transfer egg cases to a glass crystallizing dish containing control water. Discard larvae that have already left the egg cases. Add 1 ml food to each beaker before the larvae are added. Randomly allocate 12 larvae to each replicate beaker using a Pasteur pipette. Let beakers sit for 1 hr after addition of the larvae. After this period, gently immerse/place all beakers into the exposure system.
1-end	On a daily basis, check assay system and add 1.5 ml of food to each beaker
6–10	Set up schedule for auxiliary male beakers as described for day -4 to day 0 above.
20	Randomly select four replicates from each treatment, and sieve the sediment to recover larvae for growth and survival determinations. Place recovered larvae from each replicate in preashed aluminum pans and oven dry for 24 hr at 90 °C. Install emergence traps on each of the remaining reproductive replicates.
21	Weigh dried larvae to the nearest hundredth milligram. Ash the pan and larvae at 550 °C for 2 hr. Allow pans to cool to room temperature in a desiccator and reweigh. Individual larval dry weight is determined as the difference between the total mean dry weight and total mean ashed weight divided by the number of surviving larvae per replicate.
23-end	On a daily basis, record emergence of males and females, pupal and adult mortality, and time to death for previously collected adults. Each day, transfer adults from each replicate to a corresponding reproduction/oviposition (R/O) chamber. Transfer each primary egg mass from the R/O chamber to a corresponding petri dish and monitor for incubation and hatch. Record each egg case oviposited, number of eggs produced (using a direct count or the ring method [Benoit et al. (34)]), and number of hatched eggs.
30	Place emergence traps on auxiliary male replicate beakers.
33-end	Transfer males emerging from the auxiliary beakers to individual inverted petri dishes. The auxiliary males are used for mating with females from corresponding treatments for which most males have already emerged (because of protandrous emergence).
40-end	After 7 d of no emergence in a given treatment, terminate that treatment by sieving the sediment to recover remaining larvae, pupae, and pupal exuviae.

Adapted from Benoit et al. (34).

attach to the bottom and become difficult to move), and only larvae that emerge from the egg cases after this transfer should be used to initiate the test. The larvae are transferred to the exposure beakers using a Pasteur pipette and the aid of a microscope. This stage of the assay can be difficult for those who lack practice in manipulating larvae of this size (typically <1 mm), and it is recommended that the user become familiar with this procedure before initiating an assay. However, starting a test with larvae at this life stage, which is typically the most sensitive to contaminants, may yield the most accurate estimate of toxicity. To aid in the successful transfer of these small larvae, the user can periodically verify the number of larvae in the pipette by holding it up to a light and against a black background. Typically each beaker receives 12 larvae, although tests using 10 larvae have been conducted successfully (44).

A unique aspect of the life cycle test is the addition of auxiliary beakers on day 10. These beakers are necessary to supply males to compensate for the protandrous emergence pattern of *C. tentans*; without the auxiliary beakers, many of the later emerging females would not have males with which to mate, diminishing the ability to adequately assess reproduction. The auxiliary male beakers are set up following the same procedures as those described for the beginning of the test.

The primary endpoints measured in the life cycle test are survival (at 20 days and at the end of the test), growth, emergence, and reproduction. If desired, additional measurements on pupal and adult survival may be recorded. Growth and survival are determined after 20 d in the same manner as described for the 10-d test. Emergence is assessed daily over a 20–30-day period using emergence traps placed on top of each beaker (34). The emergence traps double as reproductive chambers in which mating takes place and females can oviposit. Reproductive output is determined by counting the number of eggs in each egg case; this is achieved by counting the number of eggs in five equidistant rings along the egg rope and multiplying by the total number of rings in the egg case (34). The eggs are incubated until hatch is complete (approximately 6 days); at which time, remaining eggs are counted; the difference between the number of unhatched eggs and the total number of eggs from the original count provides an estimate of percent hatch. In situations where eggs are difficult to count because of poorly formed egg cases, they may be counted directly by dissolving the egg case in dilute acid (34). The life cycle test is terminated when no hatch has been observed in the control or reference sediment for a period of 7 days. Test acceptability criteria for survival and growth and recommended criteria for emergence and reproduction are provided in Table 2.

The *C. tentans* life cycle test has been evaluated in a round-robin precision evaluation, the results of which are summarized in U.S. EPA (12), and it has been used successfully in several evaluations of contaminated sediments and water-borne exposures (25,26,34,44–46). With the length of the *C. tentans* life cycle test ranging from 45 to 60 days, this test is resource- and labor-intensive relative to the 10-d test. However, because the life cycle

test includes effects on survival, growth, emergence, and reproduction, it yields a substantial amount of information that can comprehensively evaluate potential risks of contaminated sediments to benthic invertebrates. Furthermore, a significant benefit of having detailed, synoptic information on survival, growth, emergence, and reproduction is that this can be applied in population modeling as has been done recently for both *C. tentans* and *C. riparius* (26,33,47,48).

FUTURE APPLICATIONS OF CHIRONOMUS SPP. IN SEDIMENT TOXICITY TESTING

With the availability of standard sediment testing procedures and clear guidelines for their use in sediment ecological risk assessment (ERA), recent efforts have been directed toward the development of sediment quality criteria/guidelines (SQGs) for the protection of benthic communities. A detailed description of the different types of SQGs and their basis is beyond the scope of this document; for information on SQGs, the reader is referred to Long et al. (49,50), Ingersoll et al. (8), MacDonald et al. (51), and U.S. EPA (52). Future refinements in sediment ERA, along with the development of appropriate SQGs, will continue to require sediment toxicity information, and it is likely that species of the genus *Chironomus* will continue to play a key role in this regard. Some areas for future methodological development in support of sediment ERA are briefly described below.

An area of recent activity in the field of sediment toxicology has been the development and validation of *in situ* test methods. Both *C. tentans* and *C. riparius* have played a central role in the development of *in situ* sediment testing methods (53–60). *In situ* tests involve the transplantation of organisms cultured under laboratory conditions to the field where exposure takes place in specialized exposure apparatus (53,54,59,60). In most cases, *in situ* tests have been developed and evaluated in relatively shallow habitats because of the difficulty in deploying organisms remotely in deeper waters. Although *in situ* assays involving *Chironomus* spp. have been evaluated, none have been developed into standardized protocols.

Another potentially important area for future development in sediment toxicity testing with both *C. tentans* and *C. riparius* will be in the area of endocrine disruption (ED). Few invertebrate assays are currently available to investigate sediment-based ED, but recent studies have shown that species of *Chironomus* may be useful in this application. From a physiological standpoint, *Chironomus* spp. undergoes complex biochemical changes, characterized by dramatic shifts in the titers of ecdysteroids, associated with both molting and pupation. Recent evidence indicates that the activity of certain ecdysteroids can be altered by endocrine disrupting compounds (61,62), which enhances the sensitivity of organisms during these sensitive life history stages. In this context, these developmental stages may serve as ideal focal points for the development of ED-based assays using these midges (63). Some research has already been conducted in this area. Watts et al. (43) used *C. riparius* growth and reproduction to evaluate

ethynylestradiol (E2) and Bisphenol A but found little evidence of ED. Hahn et al. (61) found that mortality of males was twice as high as females in exposures of *C. riparius* to tebufenozide and hypothesized that this could reflect ED. Several studies have used mentum deformities in *C. riparius* as an indicator of ED. For example, Meregalli et al. (64) noted increased rates of mentum deformities after exposure to nonylphenol, whereas Meregalli and Ollevier (65) found that the rate of mentum deformities after exposure to E2 was not affected. Vermuelen et al. (66) found significant mentum deformities in *C. riparius* after exposure to lead and mercury but not β -sitosterol, a known endocrine disrupting compound.

A common attribute of these studies was that ED was inferred secondarily through measurements on growth, reproduction, and deformations. No attempt was made to directly link changes in hormone levels to the biological effects observed. Recently, however, Hahn and Schultz (62) attempted to quantitatively link growth changes, determined by examining the development of imaginal discs in fourth instar *C. riparius* larvae exposed to tributyltin, to changes in the concentration of ecdysteroids in incubated prothoracic glands measured using radioimmunoassay. They showed that ecdysteroid synthesis decreased in females at 50 ng/L, with a corresponding decrease in the rate of imaginal disc development. In contrast, ecdysteroid synthesis increased in males at 500 ng/L, with a corresponding increase in the rate of imaginal disk development. These results may pave the way for the development of a standardized sediment bioassay using *Chironomus* spp. that can be applied in the assessment of sediment-associated ED.

Another potentially promising area in the development of sediment test methods with *Chironomus* spp. is the application of cell lines. In fact, epithelial cell lines have been used for over a two decades in relation to molecular-and genetic-based research with *C. tentans* (67–69). Recent studies have begun to elucidate hormone regulation and signaling pathways using these cell lines (70,71), whereas others have examined receptor binding affinity relationships using excised imaginal disks of *C. riparius* maintained in epithelial cell line media (72). The information derived from these studies may provide the necessary foundation for the development of an invertebrate-based cell line using *C. tentans* that could be applied individually to assess contaminants in aquatic environments or in conjunction with endocrine-based assays (see above) for assessing endocrine disruption.

A final area that may prove useful in the assessment of sediment toxicity is the development of behavioral endpoints. Behavioral changes often precede acute and chronic effects; yet behavioral endpoints have been sorely underused in the field of aquatic and sediment toxicology. Behavioral changes are likely to be strongly linked to acute and chronic responses (73), which enhance the ecological relevance of sediment bioassays using *Chironomus* spp. In *Chironomus* spp., activities such as ventilation, tube construction, and feeding could be developed into readily measured behavioral endpoints that may be more sensitive indicators of sediment contamination than many of

the traditional measurement endpoints (e.g., survival, growth, etc.) (74).

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CIENEGA

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Cienega is both a hydrologic and ecological term used primarily in the southwestern United States and northwestern Mexico to designate a riparian marshland. The term cienega derives from the Spanish phrase *cien aguas* (literally “hundred waters” or more figuratively, “place of one hundred springs”). The modern Spanish equivalent is *ciénaga* (“marsh, swamp”).

Most often, the term cienega refers to broad riparian areas of slowly flowing or ponded water, populated by dense vegetation. Such hydrologic conditions are unique in the Southwest because of the limited and declining number of perennially flowing stream reaches in the arid Southwest. Cienegas may also result from the discharge of springs or other conditions involving shallow or discharging groundwater. The term has also been applied to more diverse aquatic habitats throughout areas of Hispanic influence.

In South America, cienega generally refers to floodplain lakes that have open water (1). Thus, any marshy area within the Chihuahuan-Sonoran-Mohave desert region may be called a cienega. In upland environments in Arizona, the term cienega has been used to refer to marshy to bog-like meadowlands fed by seepage, precipitation,

or moving waters bordering headwater streams (1). A database maintained by the Arizona Chapter of The Nature Conservancy (TNCA) lists 160 named cienegas in the Arizona region, including two in New Mexico; 34 in Sonora, Mexico; and one in Chihuahua, Mexico. Efforts are underway to verify and characterize each site (2).

Like many other wetland habitats, cienegas have been characterized as vanishing landforms, owing primarily to certain anthropogenic factors. However, increasing awareness of the significance and uniqueness of wetlands in general has resulted in various programs to maintain and restore such areas.

HYDROLOGIC CONDITIONS

As in all wetland environments, perennial moisture at shallow soil depths, if not standing water, is requisite for a cienega. These conditions, however, are rare in arid or semiarid landscapes, where surface drainage is typically ephemeral and flows only in response to direct surface runoff. In such dry climates, the water table is generally found at considerable depth below the base of the streambed, and groundwater discharge is thereby unable to contribute baseflow. Ephemeral streams are often incised due to high loading during high intensity, short duration precipitation. Historical information indicates that arroyo cutting and incisement of many stream reaches in the Southwest began near the end of the nineteenth century from both climatic and anthropogenic changes. In contrast to typical ephemeral drainages, groundwater beneath cienegas is generally at, or very near the surface, thereby supporting baseflow and a variety of wetland species. Additionally, the cross-channel profile in cienegas is comparatively shallow and broad, sometimes extending thousands of feet across.

Cienegas can form as a result of various hydrologic, geologic, and/or ecological conditions. To sustain indigenous wetland species, cienegas require replenishment of water at rates in excess of evaporation and transpiration rates. This replenishment is supplied from shallow or discharging groundwater that occurs due to structural and/or topographic factors underlying the streambed. Hendrickson and Minckley (1) have noted various in-stream conditions that may form cienegas along stream courses. These include alternating aggradation and degradation by floods, causing alternating concave/convex stream profiling; beaver damming; slope breaks, such as at the toes of bajadas or alluvial fans; upfaulted bedrock causing localized thinning of alluvial material; channel base level adjustments with time; and impoundments caused by debris such as from slumps and landslides. Efforts to restore former cienega areas have involved constructing various types of dams and impoundments.

ENVIRONMENTAL CONDITIONS

Cienegas are wetland ecosystems and thereby represent transitional environments from terrestrial to aquatic systems. Classified wetland ecosystems require three conditions: (1) the land environment supports predominantly

aquatic flora at least periodically, (2) the substrate comprises mostly saturated soils, and (3) submergence and/or saturation of the substrate persists at least for some time during the annual growing season (3). Hendrickson and Minckley (1) defined three types of cienegas in the Southwestern United States initially based on elevation. Their elevation tiers included

- High-elevation (>2,000 m) cienegas described as marshy to bog-like alpine and cold temperate meadowlands.
- Midelevation (1,000 to 2,000 m) cienegas resulting from perennial springs and headwater streams, typically found in semidesert grasslands and evergreen woodlands.
- Low-elevation (<1,000 m) cienegas occurring as subtropical marshes in oxbows, behind natural levees, and along margins of major streams.

TNCA has noted features and characteristics common to all tiers, including groundwater at or very near the soil surface; perennial surface water; occurrence within basins, usually within unincised channels; and dominant herbaceous, perennial flora, including terrestrial, aquatic, or semiaquatic species, where scattered, less dominant woody species occur (4). These characteristics further demonstrate that cienegas meet the wetland classification criteria of Cowardin et al. (4).

Because cienegas represent comparatively lush, moist areas, they offer sharp contrast to adjacent dryland ecology. The combination of slow to stagnant water, the baffle effect of dense emergent flora, and the senescence and decay of this vegetation results in accumulation of rich organic detritus in the cienega basin. This detritus comprises a nutrient-rich substrate that enhances the overall productivity of these wetland areas.

A typical botanical progression from wettest to driest zones in a desert cienega might include emergent flora such as reeds (cattails, sedges, rushes) within the wettest areas. Salt-tolerant species such as salt grass might be found along the margins of the wetted areas, where remnant salts accumulate after evaporation. Broadleaf trees, such as various species of willow, cottonwood, and the invasive tamarisk, if present, might be found farther upland. These species adapt to less saline, less saturated soil conditions. Still farther upland, as depth increases to the water table, one might find a progression toward mesquite bosques. The availability of water and resultant luxuriance of forage in such semiarid regions often result in increases in herbivore populations (Fig. 1). In turn, this has been cited as a potential factor in the demise of these habitats (1).

Examples

Cienega Creek, Pima County, Arizona. Cienega Creek, located in southern Arizona, represents a discontinuous stream that has perennial flow along certain reaches. Perennial flow along this reach supports a rich riparian habitat that has prompted the county to designate a portion of the reach as Cienega Creek Natural Preserve. The cienega exists primarily as a result of bedrock



Figure 1. San Pedro River National Conservation Area cienega near St. David, Arizona. Note vegetation transition from sacaton grasses to senescent bullrush to woody species (photo by B. Scully).

structural control over groundwater flow. Gravimetric investigations within the Cienega Creek basin have indicated that flow occurs as a result of upwardly dipping, low permeability bedrock that results in thinning of the overlying alluvium so as to force groundwater to daylight through the streambed. These sudden changes in depth to bedrock have been associated with high-angle faulting (5,6). A longitudinal profile of this regime is illustrated in Fig. 2.

Southern Arizona. Cienegas similar to Cienega Creek have been noted historically throughout southern Arizona including the San Simon, Sulphur Springs, Aravaipa, and San Pedro Valleys. Extensive archaeological research within these areas has shown that human settlements in the vicinity of such areas date back to 8000 B.C. and use continued through recent native American cultures (1,7,8). Accounts of the Mormon Battalion tell of grassy bottomlands with "... salmon trout, up to three feet long," "... profuse and luxuriant [cane grass up to six feet high]," along the San Pedro River (9). Accounts of stream conditions in the San Pedro River near St. David, Arizona, in the late nineteenth century suggest that the channel profile was flat and marshy, in contrast to its present-day incisement (10). The incisement has been attributed to a variety of anthropogenic factors including groundwater pumping, grazing, agriculture, and removal of beaver dams. There have been also been accounts of significant changes in the overall hydrologic conditions attributed to a 1887 earthquake centered to the south in Sonora, Mexico. Reports of liquefaction, new springs, disappearance of former surface water and springs, and changes in water levels in wells have all been noted (11).

Cienega de Santa Clara, Colorado River Delta. The Cienega de Santa Clara is a remnant of the former desert estuary of the Colorado River. This nearly 15,000-acre wetland habitat occupies an oxbow area of the Colorado River. Diversions from the river now limit its replenishment; the remaining flow represents irrigation drainage pumped

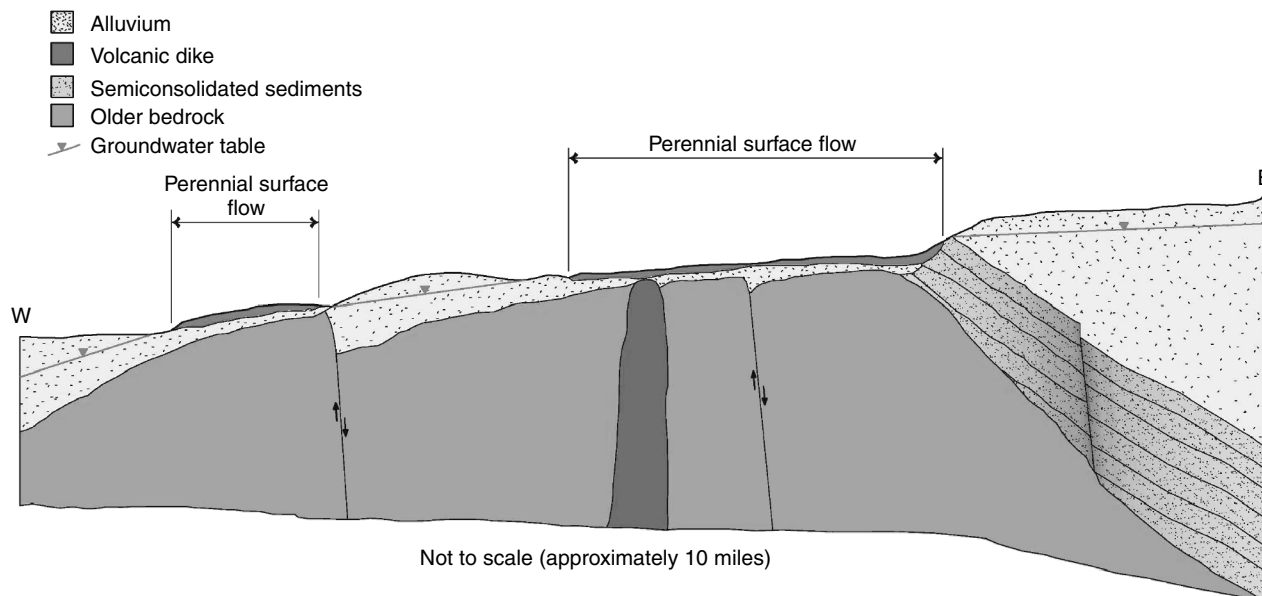


Figure 2. Conceptualization of structural components producing intermittent flow in Cienega Creek, Pima County, Arizona (5).

from agricultural land in the United States. Thus, the existence of the cienega is primarily the result of anthropogenic water management. Owing to the high rate of evaporation as well as the salinity of the influent agricultural drainage, the Cienega is mildly brackish. Recent concerns over the survivability of the habitat have arisen from consideration of diverting the irrigation outflow through a desalination plant in the United States (12). This would have the combined impact of reducing the quantity of flow and increasing salinity as a result of splicing in saline blowdown from the desalination plant (13).

RELATED TERMS

Other equivalent or related terms are billabong, bog, bottomland, carr, fen, heath, marsh, pantano, riparian, mangrove, mire, moor, muskegs, oxbow, peatland, playa, pocosin, pothole, salt marsh, slough, swamp, vernal pool, and wetland. These terms relate to various types of wetland environments, but the hydrologic conditions that form them and the ecological communities that inhabit them may differ greatly. Many terms are local and regional.

Within the Southwestern United States and Northwestern Mexico, there have been numerous such water-related terms referred to in the journals of Spanish explorers, including those of Father Eusibio Kino, a late seventeenth century missionary, who made more than 150 *entradas* ("journeys" or "reconnaissance") across the Sonoran desert during his 24-year residence. During such journeys along *El Camino del Diablo*, Kino made various references to *ciénegas*, *tinajas*, or *tanques* ("water tanks" or "jars"), and *aguajes* ("watering holes") along the way.

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TIME-AREA AND THE CLARK RAINFALL-RUNOFF TRANSFORMATION

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INTRODUCTION

In 1945, Clark introduced a synthetic unit hydrograph whose variants have been widely applied in rainfall-runoff transformation. The Clark model, designed to consider the shape of the watershed, has two components. First, the excess rainfall is translated to the outlet based on a time-area (TA) histogram. Second, attenuation due to channel storage effects is achieved by routing the time-area hydrograph into a concentrated linear reservoir. The TA method is linear and may be considered an extension of the well-known rational method. Although Clark's original concept was semidistributed over space and invariant in time, some of its recent extensions are fully distributed (e.g., (1–4)) and time variable (4).

PRINCIPLES OF THE CLARK MODEL

Let us assume that the watershed of interest, subject to a given excess rainfall, can be divided into several zones ordered by the timing of their contributions to the outlet runoff (Fig. 1). Following the convolution rule of the unit hydrograph theory, the (excess) rainfall-runoff may be transformed by

$$Q_j = \sum_{k=1}^{j_0} E_k A_{j-k+1} \quad (1)$$

where j is the time step number, Q is the runoff outlet discharge, E is the excess rainfall intensity, and A is the area of the zone. In Eq. 1, $j_0 = j$ when j is less than or equal to the time step corresponding to rainfall duration (j_r), and $j_0 = j_r$ otherwise. The application of Eq. 1 requires knowledge of the time-area histogram (TAH) which represents the time order of zone areas contributing to outlet runoff. Derivation of the TAH is a critical step in the Clark procedure and will be discussed later. Equation 1 is also known as the time-area method and constitutes the translational component of the rainfall-runoff transformation.

A fictitious linear reservoir comprises the second component of the Clark unit hydrograph which represents

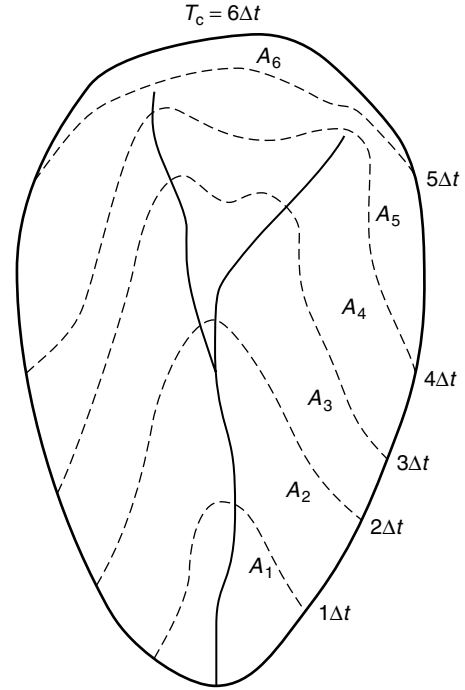


Figure 1. A watershed divided into zones (A_i) by isochrones (dashed lines).

the attenuation effects. The storage–outflow relation of the linear reservoir is usually expressed by

$$S = KO \quad (2)$$

where S is the storage, K is the storage coefficient, and O is the outflow discharge. K is a characteristic of the reservoir and has the dimension of time. The Clark hydrograph is thus obtained by routing the time-area hydrograph of Eq. 1 into the reservoir of Eq. 2. One can write the continuity equation for the reservoir as follows:

$$Q - O = \frac{dS}{dt} \quad (3)$$

where Q is the inflow discharge of Eq. 1 and t is time. For a given time step Δt , Eq. 2 expands to:

$$\frac{Q_1 + Q_2}{2} - \frac{O_1 + O_2}{2} = \frac{S_2 - S_1}{\Delta t} \quad (4)$$

in which indexes 1 and 2 refer to the beginning and end of the time step. Combining Eqs. 2 and 4 results in the following form:

$$O_2 = C_0 Q_2 + C_1 Q_1 + C_2 O_1 \quad (5)$$

where

$$\begin{aligned} C_0 &= \frac{\Delta t}{2K + \Delta t} \\ C_1 &= \frac{\Delta t}{2K + \Delta t} \\ C_2 &= \frac{2K - \Delta t}{2K + \Delta t} \end{aligned} \quad (6)$$

Note that $C_0 = C_1$ and the summation of C coefficients in Eq. 6 is equal to one.

Example

The following example illustrates the calculation of a unit hydrograph and a total hydrograph based on the Clark model. Assume that the time-area histogram of a small 1000-ha watershed is given in Table 1. Also, $K = 30$ min and $\Delta t = 15$ min.

1. Determine the 15-min Clark unit hydrograph for 1 cm unit depth.
2. Determine the discharge hydrograph corresponding to an excess rainfall intensity of 10 mm/h over the (0–30) min period and 5 mm/h over the (30–45) min period.

Solution

1. Table 2 summarizes the unit hydrograph calculation based on Eqs. 1 and 5
2. There are two possible approaches for solving this part. One is to use the unit hydrograph, as shown in Table 3. The second is to bypass the unit hydrograph and calculate the total hydrograph directly from the time-area histogram, storage coefficient, and excess rainfall hyetograph. This is shown in Table 4.

Table 1. Time-Area Histogram

Time, min	0–15	15–30	30–45	45–60
Area, ha	100	300	500	100

Table 2. Unit Hydrograph Calculation

Time, min	E^*A , ha.cm/h	Q , cms	C_0Q_2 , cms	C_1Q_1 , cms	C_2O_1 , cms	O , cms
0	0	0.00	0.00	0.00	0.00	0.00
15	400	11.11	2.22	0.00	0.00	2.22
30	1200	33.33	6.67	2.22	1.33	10.22
45	2000	55.56	11.11	6.67	6.13	23.91
60	400	11.11	2.22	11.11	14.35	27.68
75	0	0	0	2.22	16.61	18.83
90	0	0	0	0	11.30	11.30
105	0	0	0	0	6.78	6.78
120	0	0	0	0	4.07	4.07
135	0	0	0	0	2.44	2.44
150	0	0	0	0	1.46	1.46
165	0	0	0	0	0.88	0.88
180	0	0	0	0	0.53	0.53
195	0	0	0	0	0.32	0.32
210	0	0	0	0	0.19	0.19
225	0	0	0	0	0.11	0.11
240	0	0	0	0	0.07	0.07
255	0	0	0	0	0.04	0.04
270	0	0	0	0	0.02	0.02
285	0	0	0	0	0.01	0.01
300	0	0	0	0	0.01	0.01

Table 3. Total Hydrograph Derived from the Unit Hydrograph

Time, min	Discharge, cms			
	1st period	2nd period	3rd period	Total
0	0	0	0	0.00
15	0.56	0	0	0.56
30	2.56	0.56	0.00	3.11
45	5.98	2.56	0.28	8.81
60	6.92	5.98	1.28	14.18
75	4.71	6.92	2.99	14.62
90	2.82	4.71	3.46	10.99
105	1.69	2.82	2.35	6.87
120	1.02	1.69	1.41	4.12
135	0.61	1.02	0.85	2.47
150	0.37	0.61	0.51	1.48
165	0.22	0.37	0.31	0.89
180	0.13	0.22	0.18	0.53
195	0.08	0.13	0.11	0.32
210	0.05	0.08	0.07	0.19
225	0.03	0.05	0.04	0.12
240	0.02	0.03	0.02	0.07
255	0.01	0.02	0.01	0.04
270	0.01	0.01	0.01	0.02
285	0	0.01	0.01	0.01
300	0	0	0	0.01

CLARK MODEL PARAMETERS

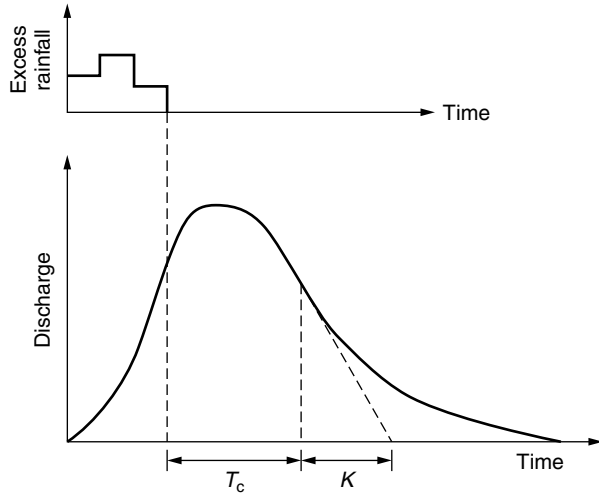
Application of Clark's model requires knowledge of some key parameters that are physically or conceptually derivable from watershed characteristics, or, alternatively, from historic rainfall-runoff data. These parameters include the time of concentration (T_c), the time-area histogram (TAH), and the linear reservoir storage coefficient (K).

Several definitions have been proposed for the time of concentration. One of the oldest definition introduces T_c as the time for (a drop of) water to travel from the most distant part of the watershed to the outlet. Some literature emphasizes that the distance has a hydraulic base, not necessarily a geometric base. Another definition deals with the wave travel time from the most hydraulically remote point to the outlet of the watershed, which is essentially the time elapsed for all parts of the watershed to contribute to the direct runoff at the outlet. In graphical analysis of a combined hyetograph-hydrograph, T_c is also defined as the time between the end of excess rainfall and the inflection point on the falling limb of the hydrograph (Fig. 2). It is assumed that the inflection point indicates the end of direct runoff. Time of concentration is essentially a function of rainfall intensity and watershed characteristics. Refer to Saghaian (5) in the *Encyclopedia of Water* for further information about time of concentration and how it differs from time to equilibrium.

A fundamental step in applying the Clark model is constructing the TAH. But first, we have to turn our attention to the concept of an isochrone. An isochrone is a contour of equal travel time to the outlet or a line connecting all watershed points that have the same travel time to a common outlet. The isochrones start and end only at the watershed boundary. The time-area histogram, also known as the time-area graph,

Table 4. Total Hydrograph Derived Directly from the Time-Area Histogram

Time, min	E_1A , ha.cm/h	E_2A , ha.cm/h	E_3A , ha.cm/h	$Q = \Sigma EA$, ha.cm/h	Q , cms	C_0Q_2 , cms	C_1Q_1 , cms	C_2O_1 , cms	O , cms
0	0	0	0	0	0	0	0	0	0
15	1000	0	0	1000	2.78	0.56	0	0	0.56
30	3000	1000	0	4000	11.11	2.22	0.56	0.33	3.11
45	5000	3000	500	8500	23.61	4.72	2.22	1.87	8.81
60	1000	5000	1500	7500	20.83	4.17	4.72	5.29	14.18
75	0	1000	2500	3500	9.72	1.94	4.17	8.51	14.62
90	0	0	500	500	1.39	0.28	1.94	8.77	10.99
105	0	0	0	0	0	0	0.28	6.60	6.87
120	0	0	0	0	0	0	0	4.12	4.12
135	0	0	0	0	0	0	0	2.47	2.47
150	0	0	0	0	0	0	0	1.48	1.48
165	0	0	0	0	0	0	0	0.89	0.89
180	0	0	0	0	0	0	0	0.53	0.53
195	0	0	0	0	0	0	0	0.32	0.32
210	0	0	0	0	0	0	0	0.19	0.19
225	0	0	0	0	0	0	0	0.12	0.12
240	0	0	0	0	0	0	0	0.07	0.07
255	0	0	0	0	0	0	0	0.04	0.04
270	0	0	0	0	0	0	0	0.02	0.02
285	0	0	0	0	0	0	0	0.01	0.01

Figure 2. Graphical representation of time of concentration (T_c) and storage coefficient (K).

is a discrete time histogram of the incremental areas enclosed by adjacent isochrones, which are apart by a finite time interval Δt (Fig. 3a). TAH represents the time distribution of sequential areas participating in surface runoff generation at the outlet. To construct the TAH, the watershed time of concentration must be divided into a number of equal time intervals, which are the travel time differences between adjacent isochrones. The isochrone spacing and construction of a TAH will be discussed later.

Some literature also introduces a concept known as a time-area diagram. A time-area diagram is the plot of area, on the y axis, whose travel time is less than or equal to a given time, shown on the x axis (Fig. 3b). From the TA diagram, one can determine the area that contributes to the outlet runoff for any given time. A TA diagram is a continuous time function. A time-area curve (Fig. 3c), or

TAC, is the derivative of a TA diagram and is equivalent to the TAH when Δt tends to zero.

The TAH can be looked upon as the basis for a unit hydrograph. When an excess rainfall of 1.0 unit depth over Δt time duration is applied to a TAH through Eq. 1, then a unit hydrograph of Δt duration is produced. For example, TAH's y axis, say, in m^2 , multiplied by 0.01 m (1.0 cm) of rainfall depth and divided by the value of Δt , say, in seconds, yields a unit hydrograph whose ordinates represent discharge in m^3/s and its volume is equal to 1.0 cm or 0.01 m across the total watershed area. When both the rainfall duration and the TAH Δt tend to zero (i.e., the isochrone time difference becomes smaller and smaller so that the TAH turns into a continuous time-area curve), an instantaneous unit hydrograph is formed.

Clark's linear reservoir storage coefficient is a physical character of the watershed that cannot be directly measured. It is, however, graphically defined as the negative of the discharge divided by the slope of the hydrograph at the point of inflection on the recession limb.

$$K = \frac{-Q}{dQ/dt} \quad (7)$$

According to Fig. 2, K is equal to the time interval between the point of inflection and the point at which the tangent to the point of inflection intersects the time axis.

An alternate way of determining Clark parameters for gauged watersheds is to use recorded rainfall-runoff to calibrate a Clark-based model for unknown parameters. For ungauged watersheds, regional empirical relations may be developed using the rainfall-runoff data of gauged watersheds in the region. In different parts of Australia, for example, relations of the following type are commonly adopted (6,7):

$$T_c = m_0 L^{m_1} S^{m_2} \quad (8)$$

$$K = n_0 L^{n_1} S^{n_2} \quad (9)$$

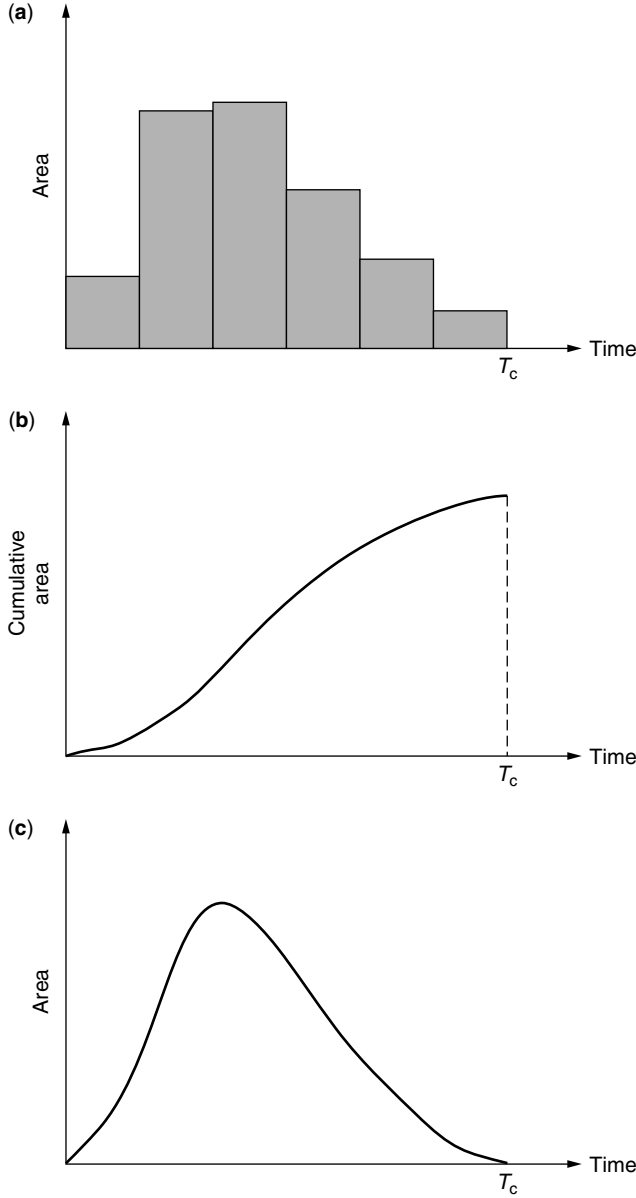


Figure 3. (a) Time-area histogram, (b) time-area diagram, and (c) time-area curve.

where L is the main stream length, S is the main stream slope, and m 's and n 's are constants for homogeneous regions.

A COMPREHENSIVE ISOCHROME SPACING METHOD

Saghafian and Julien (8) derived a general formula for the travel time from the most hydraulically remote point to the watershed outlet:

$$T_w = \int_0^L (1 - \gamma) \left(\frac{a_1}{Q_e} \right)^\gamma \left(\frac{n}{a_2^{2/3} S_0^{1/2}} \right)^{1-\gamma} dx \quad (10)$$

where T_w is the total travel time equal to the time to equilibrium, L is the total length along the hydraulically longest flow path; x is the distance measured along the

flow path; Q_e is the equilibrium discharge; a_1 , a_2 , and γ are flow cross-sectional parameters; S_0 is the bed slope corresponding to the *kinematic time to equilibrium*; and n is the Manning roughness coefficient. T_w in Eq. 10 is also a theoretical form of time of concentration. Note that travel time in Eq. 10 can also be computed for points other than the watershed outlet by changing the upper integral limit. Therefore, the travel time may be computed for all locations in the watershed using Eq. 10, and isochrones can be derived.

The equilibrium discharge Q_e passing through a given cross section at location x , is equal to the spatially integrated excess rainfall rate over the drainage area of that cross section. In mathematical terms, Q_e may be expressed by

$$Q_e(x) = \int_0^{A(x)} E dA \quad (11)$$

where $A(x)$ is the upslope drainage area or flow accumulation at distance x and E is excess rainfall intensity at any point draining to x . To solve Eq. 10, the spatial variability of excess rainfall and that of the drainage area along the hydraulically longest flow path must be known.

The total travel time, T_w , can be separated into the travel time for overland flow, T_{wo} , and for channel flow, T_{wc} . If the cross-sectional area and the hydraulic radius can be expressed by relations of the form $A_x = a_1 h^{b_1}$ and $R = a_2 h^{b_2}$, where a_1 , a_2 , b_1 , and b_2 are constants for a given cross section and h is the flow depth, then,

$$\gamma = \frac{2b_2}{2b_2 + 3b_1} \quad (12)$$

when Manning's equation is used. For overland flow and flow in wide channels, a_2 , b_1 , and b_2 equal unity and $\gamma = 2/5$ (8). If the excess rainfall intensity E is uniform in space, then $Q_e = EA$, and the term $1/E$ can be taken out of the integral. Thus, the travel time is proportional to $E^{-\gamma}$. For watersheds that have wide channels,

$$T_w = CE^{(-2/5)} \quad (13)$$

where C is a watershed hydrogeomorphic index conglomerating spatially distributed characteristics such as surface roughness, slope, flow length and flow accumulation, drainage pattern, and channel geometry. Comparing Eq. 13 with the available empirical formulas of time of concentration, one observes that most empirical formulas ignore the effect of rainfall intensity.

SIMPLIFIED ISOCHROME SPACING METHODS

Laurenson (9) proposed an early method of isochrone spacing. He assumed that the travel time from any point in the watershed to the outlet is proportional to $\sum (l_i/S_i^{1/2})$, where l_i is the length of the i th reach along the flow path of that point toward the outlet and S_i is the slope of the reach. The summation is over the entire flow path. The procedure starts by selecting a number of points

on all elevation contours distributed uniformly. The flow paths to the outlet originating from all points are plotted. The flow reaches are bounded by adjacent contours that have uniform elevation differences of, say, H . Therefore, the value of $l_i^{3/2}/H^{1/2}$ may be computed for any reach, where the slope has been substituted by H/l_i . That is to say that the travel time is proportional to $l^{3/2}$, provided that the reaches are measured between adjacent elevation contours. Once the summation values are computed for all points, they are divided by the largest summation value which represents the maximum travel time. This yields the relative travel times for the points selected. Now the isochrones can be plotted by interpolating the point values. It is easy now to develop the TAH. Geographic information systems (GIS) can be deployed to facilitate and improve the procedure. A difficulty of this method is that an independent estimate of the absolute value of maximum travel time, or the time to equilibrium, is still required for further application of the time-area method.

Pilgrim (10) argues that, based on some tracer observations in a 96-acre watershed with a dense stream network in Australia, the ratios of velocity of different reaches to that of the longest reach vary markedly with discharge. This amounts to variable distribution of isochrones corresponding to different floods. This variability is weaker for medium to high discharges observed, however. The tracing data collected by Pilgrim (10) confirmed that flood velocities tend to increase slightly in a downstream direction through the watershed, despite decreasing slope.

Some methods rely on physiographic and geomorphological parameters to determine isochrone spacing. Pilgrim (10) examined several isochrone spacing parameters versus tracer data corresponding to two discharges collected in a small watershed. Four of the parameters were as follows:

$$\sum l_i \quad (14a)$$

$$\sum \frac{l_i}{S_i^{1/2}} \quad (14b)$$

$$\sum \frac{l_i}{S_i^{1/2} R_i^{2/3}} \quad (14c)$$

$$\sum \frac{n_i l_i}{S_i^{1/2} R_i^{2/3}} \quad (14d)$$

where l_i is the length of the i th reach, S_i is the slope, R_i is the hydraulic radius, and n_i is the Manning roughness. Hydraulic radius was estimated through field observations and geometric relations corresponding to bankful discharge. The first parameter assumes uniform velocity throughout the stream reaches. It was proposed by Clark (11) but was later neglected. Other parameters consider those reach characteristics believed to represent velocity. Parameter 14b is similar to that proposed by Laurenson (9). The last two more complex parameters gave the best overall estimates, and the commonly used second parameter was poorest. The first parameter, simply derived from topographic maps, was of accuracy comparable to that of more complex parameters. These parameters give the relative location where isochrones

intersect with streams. The method must be extended somehow to overland parts of the watershed so that the isochrones can be extended to watershed boundaries. Even then, the value of time of concentration or the maximum travel time must be known so that the time-area transformation can be applied.

If no specific TAH is available, HEC (12) proposed a simple dimensionless cumulative time-area diagram as follows:

$$\begin{aligned} A^* &= 1.414(T^*)^{1.5} & \text{if } 0 < T^* < 0.5 \\ 1 - A^* &= 1.414(1 - T^*)^{1.5} & \text{if } 0.5 \leq T^* < 1 \end{aligned} \quad (15)$$

where A^* is the cumulative area as a fraction of total watershed area and T^* is the fraction of T_c . This equation has been implemented in the HEC-1 model.

Kull and Feldman (3) assumed that travel time for each cell in a watershed is simply proportional to the time of concentration scaled by the ratio of travel length of the cell over the maximum travel length, that is, the average velocity of runoff traveling from any point to the outlet is assumed uniform and constant. Each cell's excess rainfall is then lagged to the outlet by the cell's travel length. Travel time in overland and in channels follows similar proportionality, and travel length and the watershed time of concentration must be determined a priori. The method is equivalent to the distributed form of the first parameter considered by Pilgrim (10). This approach has been implemented in the HMS model (13), a new version that replaces the HEC-1 model.

TIME VARIABLE ISOCHRONE METHOD

Rainfall intensity generally varies over time, so travel times will inevitably change. That is to say that the assumption of storm-independent unit hydrograph ordinates, represented by A 's in Eq. 1, is not valid and time variability actually occurs in isochrone maps and TAHs. The use of the variable isochrone technique in rainfall-runoff transformation was first proposed by Saghaian et al. (4). In this technique, isochrone spacing and time-area histograms are updated at those time steps where rainfall intensity changes. Any TAH derived then corresponds to a given rainfall intensity and generates a partial hydrograph. The total hydrograph is derived by convoluting partial hydrographs. The numerical procedure is briefly described as follows.

The watershed is first discretized by a raster grid. The time-variable isochrone algorithm consists of four components. The first component derives terrain-based features by taking digital topographic data, usually in vector format, as input and generates raster terrain maps such as a depressionless digital elevation model (DEM), flow direction, and flow accumulation (i.e., an upslope drainage area). The temporal and spatial distributions of excess rainfall are determined by the second component, which requires soil infiltration parameters as well as the spatiotemporal variation of rainfall intensity. The third component uses maps derived in conjunction with the map of the roughness coefficient to develop differential (pixel scale) and total cumulative travel-time-to-outlet maps,

based on Eq. 10, and a time series of isochrone maps corresponding to various excess rainfall intensities. The fourth component performs TA convolution to calculate the total runoff hydrograph. This hydrograph can then be routed into the linear reservoir.

The third component is a critical part of the computations. To generate a cumulative travel-time map, isochrones of equal travel times are derived, and the areas bounded by adjacent isochrones are determined. Aggregate time-area histograms may be sufficient for uniform excess rainfall, but the spatial extent of the areas in a given travel-time range is quite important in runoff computation when dealing with spatially variable excess rainfall. In the latter case, the equilibrium discharge map of Eq. 11 is directly substituted in Eq. 10 for travel-time calculations. Then, instead of TAH, a "time-discharge" histogram (TDH) is constructed based on the equilibrium discharge map divided by isochrones. TDH itself represents the runoff hydrograph for any period of constant excess rainfall intensity.

Note that the third component must be repeated N times, where N is the number of excess rainfall intensity maps. This is equal to the number of excess rainfall intervals of constant intensity. The final output is N isochrone maps. However, in watersheds where with wide channels subject to uniform excess rainfall, we need to execute the third module once. After derivation of the TAH for any value of excess intensity (say, E_1), the TAH of, say, E_2 intensity may be easily produced by scaling the time axis by the ratio $(E_2/E_1)^{-0.4}$. Resampling of the time axis of all TAHs may be necessary to achieve a common time interval.

In the fourth component, N incremental hydrographs corresponding to N isochrone maps are determined by convolution. These incremental hydrographs, delayed by their corresponding excess intensity times, are then superimposed to yield the total hydrograph.

GENERAL REMARKS

1. One of the major advantages of the Clark model is that the temporal rainfall pattern can be accounted for. Using the newly proposed Clark-based models discussed before, the spatial distribution of rainfall may also be considered in runoff computations.
2. It is possible to derive a discharge hydrograph directly from a TAH and an excess rainfall hydrograph. This bypasses the Clark unit hydrograph and is advantageous when only a single hydrograph is to be computed.
3. Clark's original idea has been upgraded to a fully distributed time-variant approach by Saghafian et al. (4). The proposed approach can handle the spatial distribution of excess rainfall, and it can also consider the variation of travel time due to the dynamics of excess rainfall.

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STREAM CLASSIFICATION

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Stream classification is important for characterizing the features of individual rivers or sections of river and also for determining the similarity of different streams. The latter is essential to assess which rivers are suitable for extrapolating of research results from elsewhere. Various schemes have been proposed for specific disciplines and purposes. There is no universally accepted stream classification scheme because they all require modification to apply to rivers different from those used to construct the scheme. Therefore, classificatory schemes should be continuously revised and updated.

STREAM CLASSIFICATION HIERARCHY

A nested hierarchical scheme based on the channel network which places the stream within a drainage basin or watershed context is the best approach to stream classification. Table 1 defines the characteristics of, and range of scales for, each hierarchical level. Figure 1 clearly

Table 1. Hierarchical Framework of Stream Classification Proposed by References 1 and 2 and Illustrated in Fig. 1

Classification Level	Scale (River Length) ^a		Essential Features
	Small	Large	
Watershed or drainage basin	>10 ³ m	>1,000 km	Watershed/subwatershed scale boundary conditions
Valley segment	>10 ² m	100–1,000 km	Length of valley of essentially constant form and external conditions
Reach type	>10 m	10–100 km	Length of river exhibiting relatively homogeneous channel characteristics or a consistent pattern of repetitive/alternating characteristics
Channel unit or mesohabitat	>10 ⁰ m	0.1–10 km	Area of relatively homogeneous bed material, flow velocity, and flow depth
Microhabitat	10 ⁻¹ m	<0.1 km	Patch of similar flow velocity, substrate, and cover

^aScale is only indicative and must be appropriate to basin or watershed size which is highly variable.

illustrates the relationships between each classification level, where each forms the environment of its subsystem at lower levels. The smallest linear spatial scale has been taken from Frissell et al. (1) who concentrated on small mountainous forested streams. However, the size of each level increases with watershed area and runoff. Each level of the classification scheme is now discussed.

Watershed

This is the highest level of the classification scheme (Table 1) and was proposed by Frissell et al. (1) for use in bioclimatic regions. Such regions have not been defined for many parts of the world. Nevertheless, the watershed can be characterized by other general watershed characteristics, such as the Koeppen–Gieger climatic region which provides an indicator of the natural hydrology of a river. In addition, geomorphic regions have been defined for most countries and provide a good indicator of the potential energy and bedrock confinement of a river. The nature of sediment supplied to rivers can be gauged from the generalized geology. Vegetative cover and land use partly determine sediment yields which exert a strong control on stream characteristics.

Valley Segment

Following Bisson and Montgomery (2), a valley segment refers to the valley form based on the dominant types of sediment input and transport processes (Table 1). The Frissell et al. (1) classes for valley segments have been expanded to allow for the full range of conditions likely to be experienced when attempting to classify streams. *Bedrock* valleys have high sediment transport capacities because of steep slopes and close lateral and vertical confinement. *Colluvial* valleys are partly filled with poorly sorted sediment supplied from the surrounding hillslopes. *Alluvial* valleys are partly filled with sediments deposited by the present or former river and by floodplain flows. *Lacustrine* valleys refer to those superimposed on lake sediments because many rivers discharge into terminal lakes or flow for some distance over exposed lake beds. *Aeolian* valleys are those where rivers cut through, or are dissipated in, sand dunes. *Artificial* valleys are those created by human construction such as urbanization, drainage, flood mitigation, and irrigation.

RIVER REACHES

River reaches are homogeneous lengths of channel within which hydrologic, geologic, and adjacent watershed

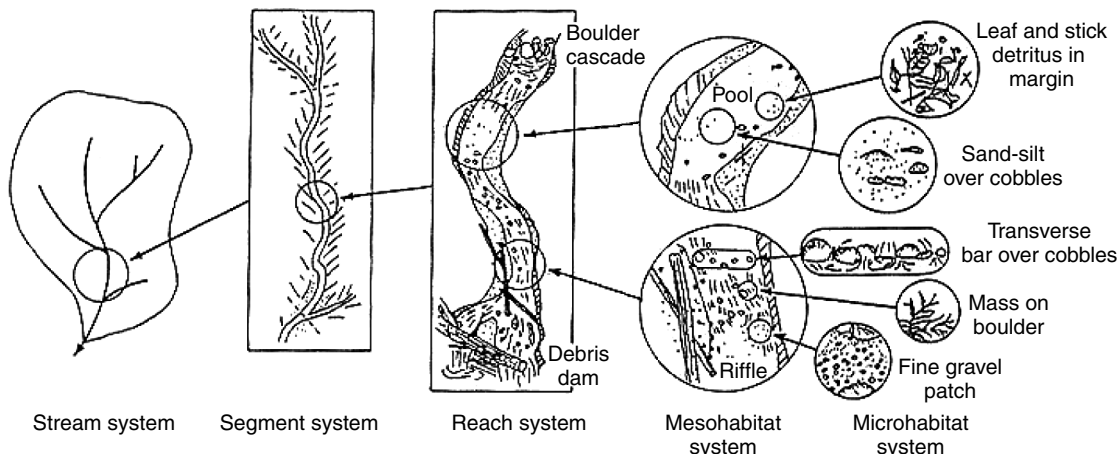


Figure 1. Hierarchy of a stream system proposed by Reference 1. The stream system is also called the watershed or drainage basin, the segment system is also called the valley segment, and mesohabitats are also called channel units (2).

surface conditions are sufficiently constant so that a uniform river morphology (3) or a consistent pattern of alternating river morphologies is produced (4). Bisson and Montgomery (2) proposed that channel reaches consist of relatively homogeneous associations of channel units, which distinguish them from adjoining reaches.

Reaches are typically 10–100 km long (Table 1). The core length of a reach is relatively easy to identify, but it is often more difficult to define precisely the boundaries of the reach due to its transitional nature (5). Bisson and Montgomery (2) proposed that provisional reach boundaries could be identified from topographic maps, vertical air photographs, and geology and soil maps, and then the channel units could be determined from selective field surveys. However, any method based on field surveys will be costly and time-consuming. There is also a problem of determining how to interpolate spatially from the sample site to the reach when the whole river is not sampled. Furthermore, a lot of detailed channel unit measurements often do not meaningfully define reach boundaries (4).

There is no agreement as to which criteria should be used to identify river reaches. Geomorphic criteria dominate most of the published schemes because reach identification is usually based on interpretation of vertical air photographs or other remotely sensed data. The schemes of References 3 and 6–10 used up to 25 different geomorphic criteria, and a similar trend is exhibited by ecologically based schemes.

Reach identification is an iterative process based on cross-checking between map and air photograph patterns and the channel units observed in the field. Therefore, periodic revisions of stream reaches should be expected as more data are collected, spatial variations in channel characteristics are better documented, and the controls on stream morphology are better understood (4).

Where appropriate, reaches can be given a formal name usually comprising three terms. The first term is based on a geographic name for a location within or near the reach; the second is a geomorphological or ecological descriptor for one of the dominant characteristics of the reach; and the third, when needed, is the term reach or zone. For example, the *Willis sand zone* on the Snowy River, Australia, is 93.5 km long and was named after Willis (the area at the New South Wales/Victorian border on the Snowy River, which is located within the reach) and the extensive sand storage that occurs in the long, relatively flat parts of this reach that are sandwiched between shorter, steeper bedrock sections (4,11).

Channel reaches can be placed into specific reach types such as those defined by Rosgen (8,9). However, Rosgen's classes must be expanded if they are to be applied universally so as to include reach types that are not currently recognized. For example, two such reach types are floodouts (12,13) and low-energy, straight channels (14,15) that are common in many parts of Australia. There are other reach types that also need to be added to the Rosgen scheme.

After identifying and naming channel reaches, each should then be allocated to one reach type for each valley segment. Each reach type can be given a simple

descriptive name based on its geomorphic characteristics, such as sandstone gorge, anabranching river, or straight, low-energy river. This is more meaningful than the alphanumeric naming system of Rosgen (8,9). For example, "sandstone gorge" is a more informative name than "A1." Correlations between rivers should be based on reach type.

Channel Units

Hawkins et al. (16) defined channel units as "...quasi-discrete areas of relatively homogeneous depth and flow that are bounded by sharp physical gradients..". Recent research has demonstrated that each channel unit exhibits different physical characteristics and can also be associated with habitat-specific assemblages of fish species (17,18). Figure 2 outlines the channel units that are commonly recognized by geomorphologists and fish ecologists as useful for stream classification. A three-tiered system was proposed by Hawkins et al. (16) so that the user can select the level of channel unit resolution appropriate for the question being addressed. At the highest level, a bipartite division of channel units has been adopted, namely, slow water and fast water (Fig. 2). Each higher level channel unit has then been subdivided into two types, which are then further subdivided on the basis of hydraulic characteristics and the principal kind of habitat-forming structure or process (Refs. 2, 16, and 19).

Channel units are defined on the basis of low flow or baseflow conditions, and slow water habitats consist of various types of pools, which are formed by either erosion (*scour pools*) or damming (*dammed pools*). For the scour pool category (Fig. 2), additional classes have been added to those of Bisson and Montgomery (2) and Hawkins et al. (16) based on Webb and Erskine (19) and are outlined in Table 2. Each scour pool is now briefly defined. A *plunge pool* is a deep pool eroded at the base of a waterfall (2). Turner and Erskine (20) coined the term "*scour hole*" for large, deep holes eroded at the boundary of a bedrock gorge with alluvium. There is no waterfall at the transition from bedrock gorge to alluvium, and hence, the holes are not plunge pools because there is no vertical drop. Furthermore, scour holes are often sand-floored rather than lined by boulders or bedrock (20). Unlike Bisson and Montgomery (2), Hawkins et al. (16), and Grant et al. (21), *step pools* are differentiated from plunge pools because of the substantial difference in the surface area and depth of the pools and in the height of the fall at the upstream end of the pool. Step pools are usually orders of magnitude smaller and have smaller falls into them. They are small pools eroded by a fall over bedrock, boulders, and log steps in steep channels and are more closely spaced than free-formed pools associated with riffles (21). *Free-formed pools* are intended to cover standard pools eroded by high flows in many alluvial channels (22). They are rhythmically spaced five to seven channel widths apart and are closely associated with riffles, which are formed by the deposition of the sediment eroded from the pools by high bankfull flows (22). *Eddy pools* are eroded by turbulent eddies downstream of obstructions and are proportional to the size of the obstruction which is usually bedrock, boulders, or large woody debris. *Trench*

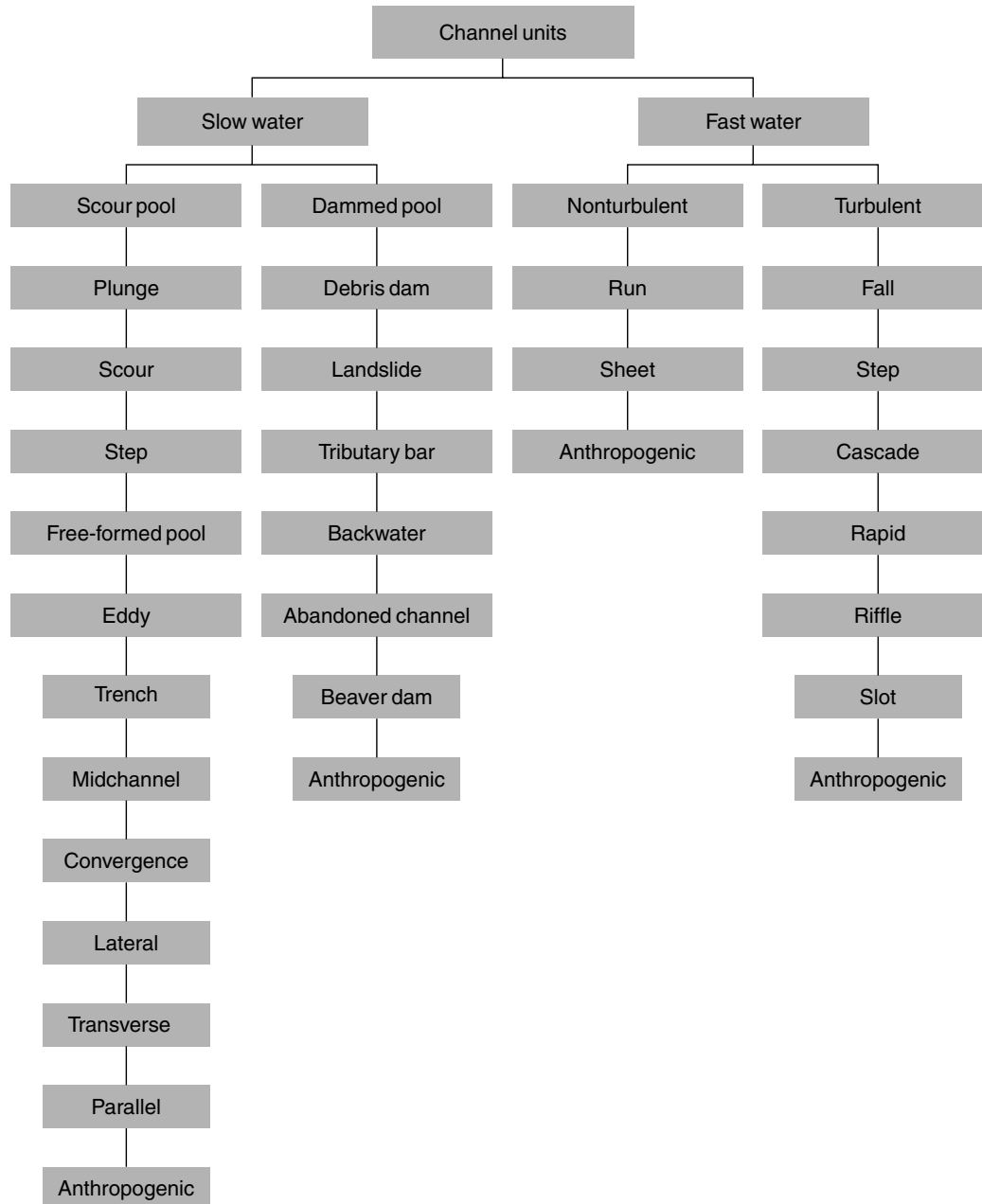


Figure 2. Channel units adapted from References 2, 16, and 19.

pools are uniformly eroded deep pools sandwiched between resistant, usually bedrock banks. They are a feature of the inner bedrock channels of Schumm et al. (23). *Midchannel pools* are formed by flow constrictions that concentrate scour in the center of a river. They differ from trench pools in that the constriction is oriented perpendicularly to the channel rather than parallel. *Convergence pools* form by the erosion of converging currents from two channels. These may be two independent channels (that is, a tributary joining the main stream) or two channels that form part of a braided or anabranching network. *Lateral* or *longitudinal scour pools* develop by erosion along or beside an obstruction or resistant material on one side of a river. They are common on bends where

the river impinges against bedrock or large bank-attached pieces of large woody debris (19). *Transverse scour pools* are formed by scour under suspended, immobile pieces of bank-attached, channel spanning large woody debris or trees aligned transverse to the flow direction (19). *Parallel scour pools* are eroded by continuous scour at the head and around both sides of bed-attached large woody debris or large boulders aligned parallel to the flow direction and not attached to the banks (19). *Anthropogenic scour pools* are included for completeness to account for scour induced by human structures.

For the dammed pool category (Fig. 2), the classes of Bisson and Montgomery (2) and Hawkins et al. (16) have been slightly expanded (Table 2). *Debris dam pools*

Table 2. Types of Pools or Still-Water Channel Units

Pool Type	Formative Process	Common Locations
Plunge pool	Large-scale turbulent scour at base of waterfall	Immediately below high outcrops of resistant bedrock or the edge of escarpments
Scour hole	Large-scale turbulent scour at transition from resistant bedrock to alluvium	Where faults or monoclines cut across a river
Step pool	Small-scale turbulent scour below bedrock, boulder, or log steps	Steep upper reaches of channels or in reaches with high loadings of large woody debris
Free-formed pool	Large-scale, rhythmically spaced turbulent scour by flood flows in gravel and/or sand-bed streams	Alluvial reaches and often associated with bends
Eddy pool	Turbulent scour by eddies associated with obstructions, such as large boulders or large woody debris	Bedrock-confined reaches, where rivers flow across resistant bedrock or floodplain reaches with large riparian trees
Trench pool	Turbulent scour at channel constrictions between bedrock walls	Gorges, bedrock-confined reaches or localized outcrops of resistant bedrock
Midchannel pool	Turbulent scour at channel constrictions oriented at right angles to the river	Localized outcrops of thin resistant bedrock that strikes at right angles to the river
Convergence pool	Turbulent scour by converging currents from two channels	Tributary junctions, multichannel reaches (braided and anabranching rivers) or where flood chutes rejoin the main channel
Lateral longitudinal scour pool	Turbulent scour along an obstruction or resistant material	Wherever a river locally impinges against large woody debris or resistant bedrock; generally in higher energy reaches than transverse scour pools
Transverse scour pool	Turbulent scour under suspended, bank-attached, channel-spanning large woody debris or trees aligned transverse to flow	Wherever a river is flanked by a riparian forest and stream power is insufficient to remove suspended timber totally
Parallel scour pool	Continuous large-scale turbulent scour at the head and along both sides of bed-attached large woody debris or boulders aligned parallel to the flow	Wherever a river is flanked by a riparian forest, and stream power is insufficient to redistribute an obstruction from the center of the channel to either bank
Anthropogenic scour pool	Small- and large-scale turbulent scour induced by an engineering structure, such as on the downstream side of a weir or dam.	Essentially random depending on location of road crossings, sites with good foundations for a structure, etc.
Debris dam pool	Impoundment by large woody debris	Wherever a river is flanked by a riparian forest and stream power is insufficient to remove obstruction totally
Landslide dam pool	Impoundment by the debris transported by a landslide	Steep upland reaches where there are high rainfall intensities and cohesive soils
Backwater pool	Pools where one channel is dammed by another or where a local channel expansion prevents active flow	Can occur throughout river systems
Disconnected pool	Impoundment of a water-table window away from the main channel by a bar of sediment	Wherever bars are well developed
Abandoned channel pool	Ponding of a deep section of a former river course	Where floodplains are well developed
Beaver dam pool	Impoundment of water by a woody barrier constructed by beavers (<i>Castor canadensis</i> ; <i>Castor fiber</i>)	Where beaver colonies are present
Anthropogenic dammed pool	Impoundment of water by an engineering structure, such as a weir or a causeway	Essentially random depending on location of road crossings, sites with good foundations for a structure

are impounded by one or more pieces of large woody debris that is anchored in the channel. *Landslide dam pools* are impounded by the debris deposited by a mass movement into the channel. Tributary-mouth bars are downstream elongated sediment bodies originating at the confluence of one, usually smaller channel with another (24). Such bars can accumulate in the channel, impounding a pool upstream (25). *Backwater pools* occur along the edges of channels at local expansions or at junctions with secondary channels. Flow separation envelopes with reverse or upstream-directed currents

are often present in the expansions. *Disconnected pools* are completely separated from the main channel at low flows by a bar or mound but are deep enough to be a window in the water table. They may be associated with secondary channels. *Abandoned channel pools* are ponded sections of a cutoff or avulsion that have no surface water connection to the main stream. *Beaver dam pools* are formed by impoundment by a woody barrier constructed by beavers (*Castor canadensis*; *Castor fiber*). *Anthropogenic dammed pools* are included for completeness to account for structure-induced impoundment.

Fast water habitats are divided into *nonturbulent* and *turbulent* channel units (Fig. 2). A *run* is a deeper, slower, less steep fast water bedform than a riffle (see below) that exhibits no supercritical flow. A *sheet* is a section of uniform water flow over smooth bedrock of variable gradient. An *anthropogenic nonturbulent unit* is one constructed by humans such as a causeway.

Turbulent fast water channel units (Fig. 2) are similar to the classes of Bisson and Montgomery (2) and Hawkins et al. (16) and are characterized by supercritical flow and hydraulic jumps sufficient to entrain air and create localized patches of white water. However, different definitions are proposed below for some of their channel units. *Falls* are high, essentially vertical drops, spanning the whole channel. The distinction between a fall and a bedrock, boulder, or log step (see below) is height. Falls are higher than the bankfull channel depth and excavate plunge pools at their bases. *Steps* are low (less than bankfull channel depth), essentially vertical drops over bedrock boulders or logs that may not span the channel. *Cascades* are steep sections composed of a series of bedrock, boulder, and/or log steps that span the channel in a staircase fashion separated by closely spaced step pools (21). More than 50% of the stream exhibits supercritical flow. *Rapids* are less steep than cascades and consequently exhibit irregular bedrock, boulder, and/or log steps that partially or fully span the channel (21). Between 15 and 50% of the stream displays supercritical flow. *Riffles* are less steep than rapids and do not display bedrock, boulder, or log steps (21). They are characterized by shallow subcritical flow, and only 5–10% of the water surface area exhibits supercritical flow (21). The chutes of Bisson and Montgomery (2) and Hawkins et al. (16) have been renamed “*slots*” because chute already has an established, alternative meaning in geomorphology, namely, a secondary flood channel across the inside of a bend or parallel to the main channel. An *anthropogenic turbulent unit* is one constructed by humans such as a dam spillway.

Microhabitats

Microhabitats are patches within a particular channel unit that have relatively homogeneous substrates, water depth, flow velocity, and cover (Table 1 and Fig. 1). No attempt is made to define classes here because of the large number of classes involved.

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COASTAL WETLANDS

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GENERAL

Coastal wetlands include a variety of wetland types such as salt marshes, bottomland hardwood swamps, fresh marshes, seagrass beds, and mangrove swamps. Although

many people think of tidal salt marshes when they hear “coastal wetlands,” there are many wetlands in coastal areas that are neither tidal nor salty. Coastal wetlands can be defined as all wetlands in coastal watersheds, that is, watersheds that drain to the ocean or to an estuary and extend landward to the limit of tidal influence (Fig. 1). Great Lakes wetlands are often considered coastal wetlands as well, but the remainder of this discussion focuses on coastal wetlands associated with the Atlantic and Pacific Oceans and the Gulf of Mexico. Coastal wetlands can be next to streams, behind barrier beaches, on deltas, in isolated depressions, or in numerous other places in coastal landscapes. Coastal wetlands such as seagrass beds also occur in bays, estuaries, or other shallow coastal waters. Whether a coastal wetland is tidal or nontidal, fresh or saline, it influences and is influenced by the entire coastal watershed.

Like all wetlands, coastal wetlands are sometimes easy to recognize, but they also can be very difficult to distinguish from uplands when obvious signs of water are absent. Coastal wetlands can occur in areas with standing water, tidal flooding, or only periodic or seasonal flooding. Coastal wetlands, like other coastal environments, are very dynamic, responding to the effects of waves, winds, currents, sea level changes, subsidence, and human alterations of the landscape and hydrology. For example, barrier beaches along the Atlantic coast of the United



Figure 1. Coastal watersheds based on U.S. Geological Survey hydrologic units.

States tend to “roll” toward the land in response to relative sea level rise (1), causing the salt marshes directly behind these beaches to be buried by the beach as new areas of salt marsh become established behind the beach on areas where wind or water carries sand. Coastal marshes around brackish ponds can change from salt marshes to fresh marshes if the pond’s connection to salt water is blocked by drifting sand or a road. Coastal fresh marshes along streams may expand or shrink depending on the changing location of the river channel.

Coastal wetlands make up about 30% of the wetlands in the continental United States, or approximately 27 million acres. The Gulf of Mexico region includes 51% of the coastal wetlands in the lower 48 states. The south Atlantic region contains another 30%, which means that approximately 81% of coastal wetlands in the continental United States are in the southeast. The entire Pacific coast (excluding Alaska and Hawaii) contains less than 2%. Most coastal wetlands (77%) occur in or adjacent to estuaries. The Ten Thousand Islands estuary in southern Florida, known for its mangroves and extensive areas of forested, shrub, and fresh marsh in the Everglades, contains the largest acreage of coastal wetlands in the United States, followed by the Mississippi Delta estuary, and the Albemarle/Pamlico Sound estuary (2).

Only approximately 18% of coastal wetlands are truly estuarine (having water with a salinity of at least 0.5 ppt) and only about 25% are tidal. The most common coastal wetland type is forested and scrub-shrub, which comprises 63% of the coastal wetlands in the continental United States. The next most common coastal wetland type is fresh marsh (16%), followed by salt marsh (15%) and tidal flat (4%) (2).

ECOLOGICAL IMPORTANCE

Coastal wetlands are among the most productive ecosystems in the world, comparable to tropical rain forests and coral reefs. An immense variety of microorganisms, plants, insects, amphibians, reptiles, birds, fish, and mammals can be part of a wetland ecosystem. Coastal wetlands can be thought of as “biological supermarkets.” They provide large volumes of food that attract and support many animals. The combination of shallow water, high levels of nutrients, and high primary productivity is ideal for the development of organisms that form the foundations of the foodweb, supplying food for many species of fish, shellfish, insects, amphibians, birds, and mammals. A small percentage of wetland vegetation is eaten directly by fish and invertebrates, but most is decomposed by bacteria, producing an organic soup that feeds organisms such as amphipods, shrimp, crabs, snails, shellfish, and finfish. These organisms in turn support a broad food chain that includes a variety of fish, birds, and mammal populations such as deer, muskrat, and manatee.

Coastal wetlands are extremely important to birds, which use them for feeding, resting, and/or nesting. Major migratory flyways follow coastal wetlands along the Pacific and Atlantic Oceans. Many shore birds are wetland-dependent, including gulls, terns, plovers, and sandpipers. Waterfowl such as bay ducks (canvasback,

redhead, etc.) and sea ducks (bufflehead, harlequin duck) are dependent on coastal wetlands, and some declines in coastal waterfowl populations are thought to be linked to the decline in coastal wetlands (3).

Coastal wetlands are also important nurseries for juvenile marine organisms. The shallow water and often-abundant plant growth are ideal for sheltering animals from predators until the animals are big enough or fast enough to avoid predators outside the wetland. Some fish, including species of herring and bass, use coastal wetlands as both a spawning area and nursery, whereas others, such as some species of shrimp and the ladyfish (a popular sportfish) spawn in the open ocean, leaving the juveniles to migrate via coastal currents to coastal wetland nurseries. Even adult fish often use coastal wetlands as refuge from larger predators or environmental stresses such as storms or floods.

Coastal marshes and estuaries provide essential habitat for over 75% of the fish caught commercially and recreationally in the United States (4). Popular species such as spotted seatrout, scup, butterfish, mullet, spot, salmon, gag grouper, Pacific herring, white grunt, summer flounder, menhaden, pink shrimp, spiny lobster, Atlantic croaker, and blue crab all use coastal wetlands as juveniles or adults for feeding and refuge. Bluefish and striped bass, important recreational species, depend on salt marshes for the small fish that are an important part of their diet. Menhaden, one of the most important commercial species landed in the United States, has a particularly strong tie to salt marshes and their detrital food chain. All of these species, and many others too numerous to list here, depend on healthy abundant coastal wetlands to support their populations.

Coastal wetlands are also important for keeping coastal waters clean. Coastal ecosystems receive virtually all the water flowing off the upland landscape—water that may contain fertilizers, pesticides, sewage, sediment, or toxic chemicals. Coastal wetlands filter out sediment and some chemicals, reducing the amount of pollution that washes into bays and the ocean.

TRENDS

Coastal wetlands are among the most imperiled ecosystems in the world. By the mid-1970s, over half of all salt marshes and mangrove forests present in the precolonial United States had been destroyed (5). California, a large coastal state, has lost over 90% of its wetlands. Florida and Louisiana, two coastal states with the greatest acreage of wetlands, have lost about half of their original wetland area. Louisiana alone is losing between 16,000 and 25,000 acres of wetlands a year, which is the highest sustained wetland loss rate in the country (6). Most of that loss is occurring in coastal areas.

No long-term national data set is available for coastal wetlands, although some programs, such as the U.S. Fish and Wildlife’s National Wetlands Inventory (NWI), track estuarine wetlands and the U.S. Department of Agriculture’s Natural Resources Inventory (NRI) has been used to estimate loss of tidal wetlands (7) and wetlands in coastal counties (8). Coastal wetland losses were estimated

by Gosselink and Bauman (9) as approximately 20,000 acres per year from 1922 to 1954 and approximately 46,000 acres per year from 1954 to 1974. NRI data for the period between 1982 and 1987 indicate that tidal wetland loss on nonfederal lands was approximately 20,000 acres per year. However, tidal wetlands make up only about 25% of all coastal wetlands, so the total coastal wetland loss for that time period would have been greater. Most recently, Brady and Goebel (8) estimate wetlands losses on nonfederal land in coastal counties from 1992 to 1997 to have been about 24,000 acres per year. The net loss of wetlands in coastal counties was nearly three times the net loss of wetlands from inland counties. Nearly all of these losses were to freshwater nontidal coastal wetlands.

According to the NWI, the rate of wetland loss of all wetlands nationally has decreased from 458,000 acres per year from the mid-1950s to mid-1970s (10) to approximately 58,000 acres per year from 1986 to 1997 (11). While the national wetland loss rate has decreased to less than one-eighth of what it once was, the rate of coastal wetland loss appears to be decreasing much less (about one-half of what it was) and much more slowly.

Many factors are responsible for coastal wetland loss. In the early part of this century, coastal wetlands were drained and used for farming or grazing. More recently, coastal wetlands have been filled or dredged for roads, houses, golf courses, marinas, and other development. In some parts of the world wetlands are being destroyed for aquaculture. In other places, such as Louisiana, wetlands are being lost to open water due to a combination of factors including canals dredged through the marshes, dams on the Mississippi River reducing sediment to the marshes, land subsidence, and sea level rise. Even wetlands that are not actually filled or dredged are becoming degraded due to pollution, changes in water flows, and invasion by weeds or other non-native plants and animals.

Coastal wetland losses can be directly traced to population pressures and other human changes occurring along the coast. Coastal populations have increased steadily since 1970. Currently, over half the population of the United States lives in coastal counties, at densities about five times greater than those of noncoastal counties (12). This trend of people moving to coastal areas is expected to continue in the coming decades. Expanding populations place enormous pressures on existing natural resources, particularly wetlands, which are very vulnerable to changes in water flow, pollution, and habitat fragmentation. During the period from 1992 to 1997, nonfederal land in coastal counties contained 20% of the wetlands in the continental United States but experienced 31% of the wetland losses. Sixty-six percent of wetlands lost in coastal counties was the result of development while only 41% of wetlands lost in inland counties was due to development (7).

Coastal wetland loss from anthropogenic factors is magnified by global considerations such as sea level rise. Along much of the U.S. coast, relative sea level is rising at a rate of between a few inches to a foot or more per century (1). In fairly flat coastal plains,

a few inches of rise in sea level can mean that the coastline moves hundreds of feet inland. Coastal wetlands can and do move inland with rising sea level, but in developed areas roads, houses, parking lots, and other human structures interfere with this natural migration of coastal habitats.

Fortunately, the importance of coastal wetlands and their vulnerability have come to the attention of scientists, governments, and nongovernment organizations. Research into the restoration of coastal wetlands and on-the-ground application of this research have yielded promising results, allowing for the restoration of some types of coastal wetlands. Through programs like those funded under the Coastal Wetlands Planning, Protection, and Restoration Act, federal and state agencies in the United States are working together to reduce the loss of wetlands in particularly fragile areas such as coastal Louisiana. And in many places, local governments are developing land use plans to protect coastal wetlands while nongovernmental organizations develop support for these programs and advocate for additional funding for restoration. With these efforts as a starting point and additional protection and restoration efforts in the future, coastal wetlands will continue to provide enormous benefits to the ecology and economy of the world.

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FRESHWATER COLLOIDS

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INTRODUCTION

Colloids are defined on the basis of size; at least one dimension is in the range 1 nm–1 μm (10^{-9} – 10^{-6} m). Below 1 nm is the dissolved or solution phase, and above 1 μm is the particle phase. In practice, this distinction is usually made by operational methods such as ultrafiltration and filtration (1). Other techniques such as centrifugation have also been used as separation techniques. Although useful and widely used, this operational definition is inadequate for a number of reasons. First, there are practical considerations such as experimental artifacts due to filtration and mass-based, rather than size-based, separation by centrifugation. Second, the terminology is not compatible with thermodynamic conceptions according to Stumm and Morgan (2), who defined colloids as species for which a chemical potential could not be defined. Gustaffson and Gschwend (3) delimited colloids from both the dissolved phase and the particle phase, based on the chemical and hydrologic properties of the system.

Dissolved and colloid species were distinguished on the basis that the colloid forms a distinct chemical phase whose interactions with other components is different from solution phase interactions. The distinction between colloid and particle phases was based on the idea that colloids have a tendency to remain in the water column whereas particles tend to settle to the sediments.

As noted, this distinction is in part based only on size and also includes colloid chemistry and mass and hydrologic conditions. In this conception, a colloid may be so defined in one waterbody but may not be classed as a colloid in another water-body. This is similar to the distinction made elsewhere in the literature (4,5), where the behavior of colloids in surface waters is dominated by aggregation and the behavior of particles is dominated by aggregation. In porous media, colloids and particles may also be governed by retention and migration in the pores, respectively (4,6). This idea of settling particles and nonsettling colloids has important implications for their transport and fate in the environment.

PHYSICAL, CHEMICAL, AND MICROBIOLOGICAL PROPERTIES OF COLLOIDS

To understand the role and properties of colloids (aggregation, sedimentation, pollutant binding), it is essential to understand that their nature is essentially cross-disciplinary, requiring knowledge of their chemical, physical, and microbiological properties. A number of the physicochemical properties of colloids are important in this respect: they include size, shape, density, chemical composition, and surface charge. Activities of microorganisms are also important in producing and removing colloids from freshwaters and in binding trace pollutants. Colloids are also not 'pure' compounds but are intrinsically polydisperse (containing a range of sizes) and chemically heterogeneous. This inhomogeneity in size and chemistry reflects the wide variety of different sources and sinks for forming and eliminating of colloids. For example, colloids are produced by the weathering of rocks and soils and by the death, lysis, and degradation of biological cells. Additionally, colloids are destroyed by their aggregation into larger particles and sedimentation and by their use as a food source by other organisms.

Typically, freshwater colloids consist of biological, organic, and inorganic materials. These include microorganisms such as bacteria and their debris; humic substances, which are the degradation products of plants and other biota; microbial exudates, primarily polysaccharides and proteins; and weathering products such as clays and oxides. Colloids are not pure phases of these different materials, but the phases are intimately linked.

Sampling, Fractionation and Analysis

Although colloids are intimately linked mixtures of phases, they are very unstable and easily modified. Indeed, even the act of sampling may alter their composition (1,6). This instability is linked to aggregation, microbial growth and death, modification of organic matter, sorption to container walls, changes in solution conditions, and temperature. Sampling and analytical procedures must therefore be carefully chosen, the prime requirement is that they are as minimally perturbing as possible. Ideally, analysis would be performed *in situ*, although this can rarely be achieved in practice. The sampling step itself should be quick with a minimum of handling, storage, and transport. Full details of sampling procedures are given in the two previous references.

Fractionation, usually performed by size, is often required to reduce the complexity of colloids. A number of methods have been used. The main method is filtration, although centrifugation, field-flow fractionation (FFF), and others are now employed. Again, the main criterion for successful use is nonperturbation of colloids during fractionation, one that is rarely, if ever fully met.

A number of standard techniques exist to characterize the microbiological (e.g., flow cytometry) and bulk chemical composition of colloids (e.g., inductively coupled plasma mass spectrometry, ICP-MS), although these are outside the scope of this paper. Other techniques of great importance for the direct analysis of freshwater colloids include electron and force microscopy and various light

scattering techniques. Each technique has its own advantages and limitations for example, transmission electron microscopy allows direct visualization of nanometer-scale material and the analysis of major elements on a single particle basis. However, sample preparation may be problematic, despite the advances made in the past 10 years. In addition, quantification of results is difficult. Due to the different strengths of each technique and the complexity of the colloids, coupling of two or more methods on-line is advantageous (e.g., FFF-ICP-MS) as is the use of a number of complementary techniques to analyze the same sample.

ENVIRONMENTAL IMPORTANCE

Freshwater colloids have been studied primarily because of their role in chemically binding trace pollutants (metals and organics) and thus altering the transport and bioavailability of pollutants. Additionally, they have been studied in relation to water treatment processes and as a food source for aquatic microorganisms. The physical and chemical nature of colloids means that they can bind a large, often dominant, fraction of trace pollutants (7). In particular, they have high surface area. Consequently, a large number of functional groups, which can bind trace metals, are presented to the solution. Additionally, the functional groups themselves strongly bind the metals. Although less well-developed experimentally and theoretically, the binding of organic pollutants by colloids has also been investigated. It is generally thought that the organic component of colloids controls binding.

The chemical binding of metals by colloids has two implications for metal biogeochemistry. First, uptake and effects of the metals by biota are modified, and second, environmental transport of the metals is affected.

Colloids, it is thought, act to reduce biotic uptake to aquatic organisms such as microorganisms and fish. The widely used Free Ion Activity Model (8) assumes that bioavailability is directly related to the free ('dissolved') metal ion: related models (9) have explained reduced uptake by the decreased diffusive mobility and chemical lability of the colloid-bound metal. Despite the successes of these and other models (such as the Biotic Ligand Model), much remains to be understood about the interrelationships between colloids and biota and subsequent effects on metal bioavailability and toxicity.

Colloids affect transport and cycling of trace metals through a number of mechanisms. As mentioned in the definition, one definition of colloids is based on their transport behavior in water. Colloids are small enough that mixing processes maintain them in the water column, whereas particles are large enough so that they sediment out of the water column. This simple mechanism provides an important method for the geochemical fractionation of associated pollutants in surface waters. In lakes, settling of particles is one of the main 'self-purification' mechanisms for removing metals from the water column (10). Additionally, in groundwater, the flow of colloids through porous media may govern the transport of associated pollutants.

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CULVERT DESIGN

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A culvert (Fig. 1) is a conduit placed under a road to convey surface water from one side to another. In addition to its hydraulic function, it must also carry construction and highway traffic and Earth loads; therefore, culvert design involves both hydraulic and structural design. Culvert design involves selection of the type of culvert and size to pass the design discharge without overtopping of the road and without erosion on either end of the culvert. For economy and hydraulic efficiency, engineers should design culverts to operate with the inlet submerged during flood flows, if conditions permit. Culverts are constructed from a variety of materials and are available in many different shapes and configurations. When selecting a culvert, engineers should consider roadway profiles, channel characteristics, flood damage evaluations, construction and maintenance costs, and estimates of service life. The most common shapes for culverts are circular, pipe-arch and elliptical, box (rectangular), modified box, and arch. Common culvert materials include concrete (reinforced and nonreinforced), steel (smooth and corrugated), aluminum

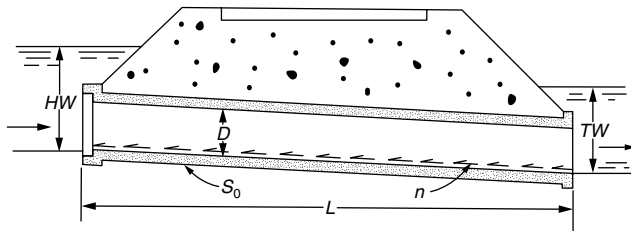


Figure 1. A typical culvert with submerged inlet and outlet.

(smooth and corrugated), and plastic (smooth and corrugated). The selection of material for a culvert depends on several factors that can vary considerably according to location. One should consider the following groups of variables: (1) structure strength, considering fill height, loading condition, and foundation condition; (2) hydraulic efficiency, considering Manning's roughness, cross-section area, and shape; (3) installation, local construction practices, availability of pipe embedment material, and joint tightness requirements; (4) durability, considering water and soil environment (pH and resistivity), corrosion (metallic coating selection), and abrasion; and (5) cost, considering availability of materials. Do not base culvert material selection solely on the initial cost. Replacement costs and traffic delay are usually the primary factors in selecting a material that has a long service life for a culvert design.

Energy is required to force flow through a culvert. This energy takes the form of an increased water surface

elevation on the upstream side of the culvert. The depth of the upstream water surface measured from the invert at the culvert entrance is generally referred to as headwater depth (HW in Fig. 1). The flow through a culvert can take different forms and is a function of several variables, such as cross-sectional size (e.g., inside diameter D in Fig. 1) and shape, bottom slope (S_0), length (L), conduit roughness (usually in terms of Manning's n), entrance and exit design, and water depths at the upstream (headwater) and the downstream of the culvert (tailwater, TW from outlet invert in Fig. 1).

Both entrance flow and exit flow can be broadly classified as submerged (e.g., in Fig. 1) or free. Two types of flow controls are in culverts: inlet control and outlet control. For inlet control (Fig. 2), the control section is located at or near the culvert entrance, and the discharge through the culvert is dependent only on inlet geometry (size, shape, entrance type) and headwater depth (HW) because the conduit can convey more discharge than the inlet will allow. For the inlet control, if the outlet is not submerged, most flow in the conduit is open channel flow; if the outlet is submerged, part of the flow near the entrance can still be open channel flow, but flow near the outlet is pressurized pipe flow. For outlet control (Fig. 3), the control section is at or near the culvert outlet, and the discharge through the culvert is dependent on all hydraulic factors upstream from the outlet. For example, flow condition in Fig. 1 is an outlet control, and all hydraulic factors HW , TW , L , D , n , and S_0 affect discharge through the culvert.

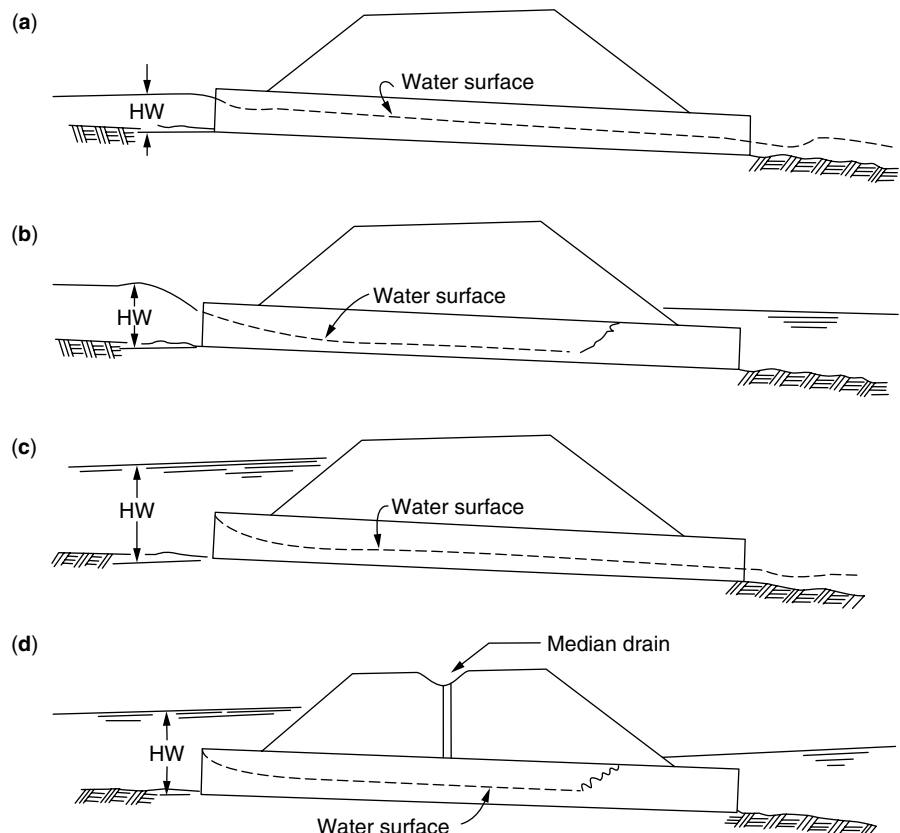


Figure 2. Types of inlet control for a culvert: (a) inlet and outlet unsubmerged; (b) inlet submerged, outlet unsubmerged; (c) inlet submerged only; and (d) inlet and outlet submerged [from Normann et al. (1)].

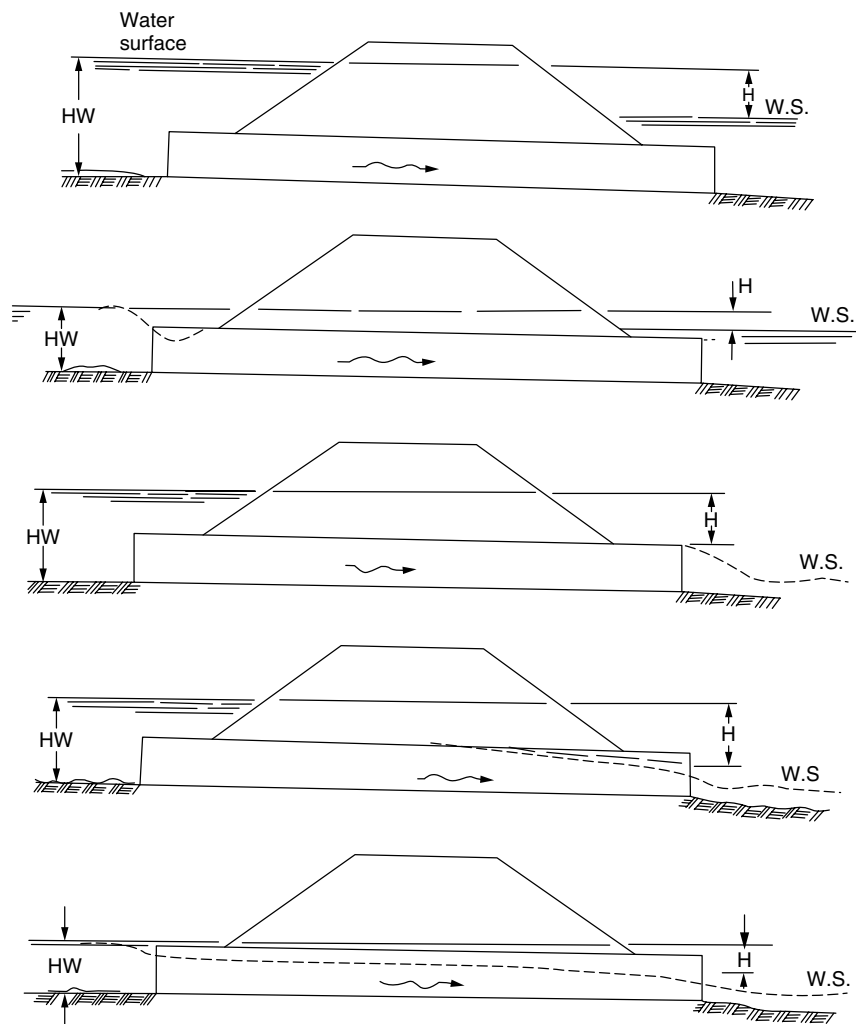


Figure 3. Types of outlet control for a culvert [from Normann et al. (1)].

The flow velocities from a culvert are likely to be higher than in the channel because a culvert usually constricts the available channel area. These increased velocities can cause streambed scour and bank erosion in the vicinity of the culvert outlet. Therefore, two basic culvert design criteria exist: allowable headwater (HW) and allowable velocity. Allowable headwater usually governs the overall configuration of the culvert. However, the allowable outlet velocity is the governing criterion in the selection and application of various downstream fixtures and appurtenances to slow the flow from the culvert. An engineer may improve culvert hydraulic capacity by selecting appropriate inlets (e.g., projecting, mitered, straight, flared, and parallel wingwalls). Many different inlet configurations including both prefabricated and constructed-in-place installations can be used by the engineer. When selecting various inlet configurations, one should also consider structural stability, aesthetics, erosion control, and fill retention.

The culvert design process typically includes the following basic steps:

1. Define the location, orientation, shape, and material for the culvert to be designed. One may consider more than a single shape and material.
2. With consideration of the site data, establish allowable outlet velocity and maximum allowable headwater depth.
3. Based on design discharges from hydrologic study of upstream watershed and associated tailwater levels, select an overall culvert configuration including culvert hydraulic length, entrance conditions, conduit shape, and material.
4. Perform hydraulic analysis to determine headwater depth and outlet velocity by using computer software or monographs.
5. If computed results do not satisfy the specified values at step 2, optimize the culvert configurations, and go back to step 3. Sometimes it may be necessary to treat any excessive outlet velocity separately (additional energy dissipation structures) from the headwater requirement.

Culvert design typically requires complex hydraulic calculations, especially to appropriately incorporate energy losses caused by entrance, friction, and exit. Equations, tables, and charts for culvert design can be found in many textbooks (2,3) or design manuals (1,4–6).

Figure 4. Culvert calculator in CulvertMaster (7).

Many computer models, e.g., HY8 by the Federal Highway Administration (FHWA) of the U.S. Department of Transportation, "CulvertMaster" developed by Haestad Methods, Inc. (7), and CULVERT2 (English units) and CULVERT3 (metric units) by the California Department of Transportation, are also available. Application of computer software is recommended because of complexity in culvert hydraulic computations. One must choose the shape and profile and establish the culvert dimensions by iterative application of the computation procedures until headwater and outlet velocity are reasonable. Sometimes it is possible to select culverts consisting of more than one box in wide channels to convey design discharge. The Culvert Calculator in CulvertMaster can solve for size, discharge, and headwater elevation for a culvert. The Calculator organizes information for a culvert into seven areas (Fig. 4): solve for, culvert, section, inlet, inverts, headwater elevation, and exit results. Figure 4 shows an example for sizing of a single, circular, concrete culvert (roughness $n = 0.13$) with square edge entrance and design discharge of $60 \text{ ft}^3/\text{sec}$ and 36 ft long. The result for the culvert size is 36 inches; flow has inlet control because the headwater elevation for inlet control is greater than the one for outlet control.

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DILUTION-MIXING ZONES AND DESIGN FLOWS

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INTRODUCTION

The possibility that a pollutant will be discharged accidentally or intentionally into a stream/river is a matter of constant concern to those diverting and using water from streams and rivers. It is essential for water resources planners and managers to understand the physical processes involved in a stream/river prior to the implementation of any water resources projects. When a pollutant is discharged into a stream/river, it is subjected to initial dilution immediately due to the density difference between the receiving water and the pollutant.

The longitudinal, vertical and two-dimensional profiles of pollutant concentration indicate the density gradient between the pollutant and the receiving water. After initial dilution, the processes are governed by advection, reaction, and dispersion phenomena that tend to modify the initial pollutant concentration.

Advection phenomena represent the downstream transport of a discrete element of the waste load by the stream flow.

The reaction phenomena represent the decay of biodegradable materials in the waste under the action of naturally occurring bacteria in the stream. The dispersion phenomena represent that under the influence of turbulence, eddy currents, and similar mixing forces, a discrete element of the waste load tends not to remain intact, but mixes with adjacent upstream and downstream elements. In rivers and streams, the influence of dispersion phenomena is usually relatively small compared with advection and reaction phenomena; however, it can be important in some circumstances.

When a slug load results from a spill or accidental dump, dispersion effects can have an important influence on resulting peak concentrations, particularly at longer distances from the point of discharge (Fig. 1). Intermittent discharges, such as storm runoff, are also influenced by dispersion. However, for continuous discharges (e.g., from wastewater treatment plants) and steady-state flow conditions, dispersion effects are usually insignificant (Fig. 2).

As a result of these processes, the pollution level downstream from a wastewater disposal site varies in time and space. In the following sections, the importance of diluting pollutants downstream of their discharge based on various physical processes, the mixing length of pollutants (mixing zone), and design flows have been described.

IMPORTANCE OF DILUTION

The purpose of dilution is to reduce the concentration of pollutants in discharges below applicable standards. All

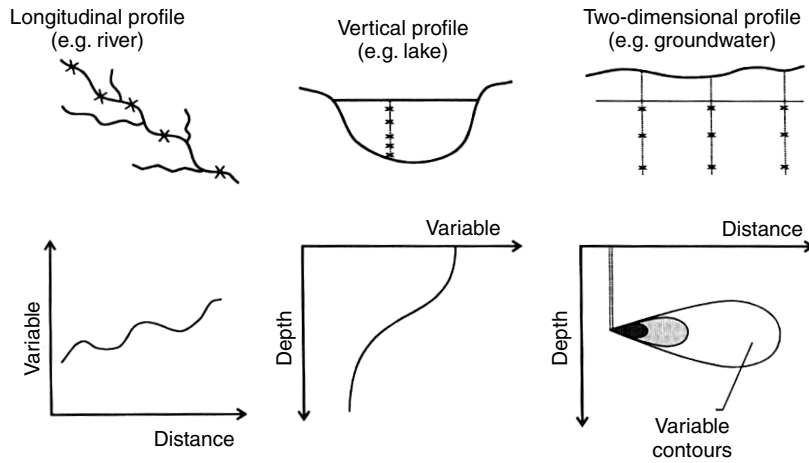


Figure 1. Spatial movement of water quality variables.

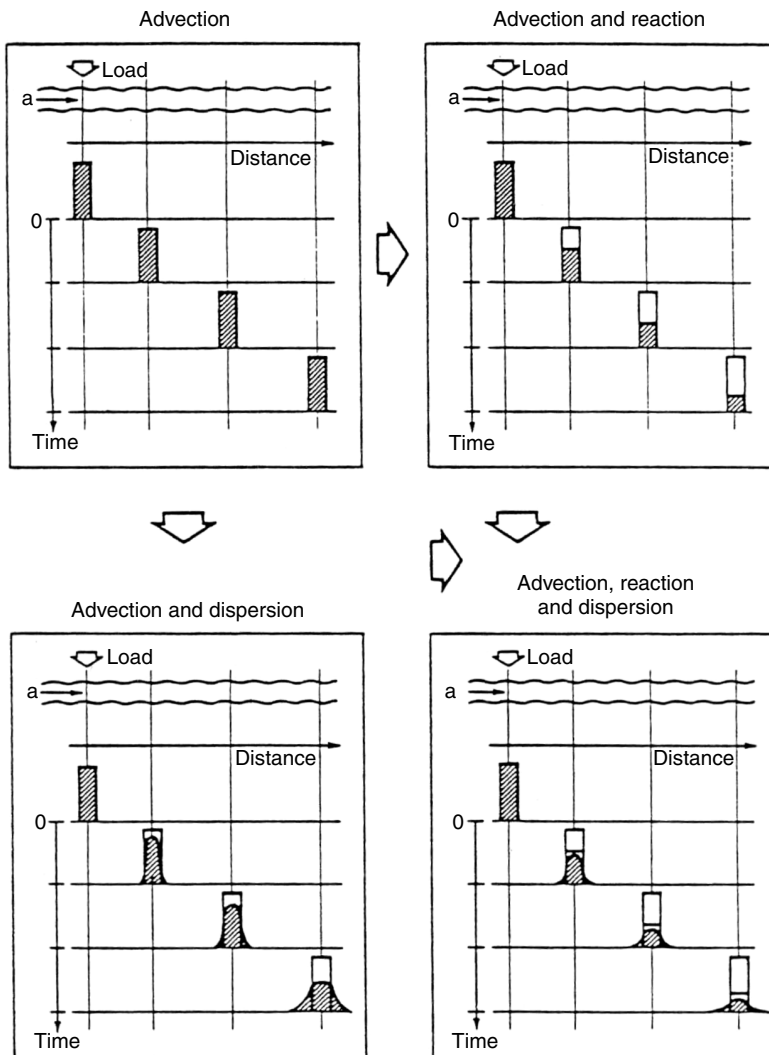


Figure 2. Transport mechanism for waste loads.

applicable water quality standards and nondegradation limits must be met at the end of a mixing zone. Through the first decades of the twentieth century, wastewater disposal practices were based on the premise that “the solution to pollution is dilution” (1). Dilution was considered the most

economical means of wastewater disposal and as such was considered good engineering practice. It is one of the main processes for reducing the concentration of substances away from the discharge point. Dilution is more important for reducing the concentration of conservative substances

(those that do not undergo rapid degradation, e.g., metals) than for nonconservative substances (those that do undergo rapid degradation, e.g., some organic substances).

Although dilution is a powerful adjunct to self-cleaning mechanisms of surface waters, its success depends upon discharging relatively small quantities of pollutants into large bodies of water. Growth in population and industrial activity and attendant increases in water demand and wastewater quantities preclude the use of many streams for diluting raw or poorly treated wastewaters or pollutants. In the United States, legal constraints further limit using waterbodies for wastewater dilution (1). Under present regulations, maximum allowable loads are set independently of dilution capacity. Only when the standard maximum loads result in violating in-stream water-quality standards, is dilution capacity considered, and then the increment of treatment necessary is determined.

To understand dilution phenomena due to increasing pollutant load in streams/rivers, different scenarios were obtained by plotting the changes in discharge versus simultaneous changes in the concentrations of various substances (2). As shown in Fig. 3, different curves may represent timescales that vary from a single storm to that of several years' duration. In the figure, curve (1) shows a general decrease in concentration with discharge, which implies increasing dilution of a substance introduced at a constant rate (e.g., major cations, particularly when concentrations are high). This is also characteristic of point source discharges such as municipal sewage and many industrial point sources. Curve (2) shows a limited increase in concentration generally linked to the flushing of soil constituents (e.g., organic matter, nitrogen species) during runoff. Curve (3) is basically the same as curve (2), but a fall in concentration occurs at very high discharges indicating dilution of soil-runoff waters. Curve (4) shows an exponential increase in concentration; this occurs for total suspended solids and all substances bound to particulate matter. The curve represents the increase in particulate matter due to sheet erosion and bed remobilization. Substances bound to such particulate include phosphorus, metals, and organic compounds, predominantly pesticides and herbicides. Curve (5) in the hysteresis loop is observed as time is introduced as an additional parameter to the sediment discharge

relationship shown in curve (4). Such patterns can be seen for total suspended solids, dissolved organic carbon, and sometimes nitrate. The peak in sediment concentration occurs at X (advanced) before the occurrence of the peak discharge at Z. Curve (6) indicates a water source to the river of constant, or near constant concentration (e.g., chloride in rainfall, groundwater, and outlet from a lake).

From Fig. 3, it is found essential to determine the stream/rivers dilution capacity prior to the development of mathematical modeling and water quality simulations.

The dilution capacity of the receiving water can be defined as the effective volume of receiving water available for diluting the effluent. The effective volume can vary according to in-stream discharge and any other input source of discharge. In streams/rivers, in particular, the effective volume is much greater during the monsoon than during a lean period. It is important to consider concentrations of substances in the worst case scenarios (usually lean flow in summer, for example, when pollutants might be carried further into a sensitive location) when calculating appropriate discharge content conditions. The dilution capacity of a stream can be calculated by using the principles of mass balance. If the volumetric flow rate and the concentration of a given material are known in both the stream and waste discharge, the concentration after mixing can be calculated as follows:

$$C_s Q_s + C_w Q_w = C_m Q_m \quad (1)$$

where C represents the concentration (mass/volume) of the selected material, Q is the volumetric flow rate (volume/time), and the subscripts s , w , and m designate stream, waste, and mixture conditions.

MIXING ZONES

Wastewater effluents are discharged into streams and rivers to minimize any potential adverse effects on river water quality by suitable siting of outfalls. The stream zone between the outfall and the nearest cross section of uniform concentration distribution is known as a "mixing zone." The mixing of effluent with the river water takes place by the "jet effect" in the near-field region as shown in Fig. 4 (MixZon Inc., 2003).

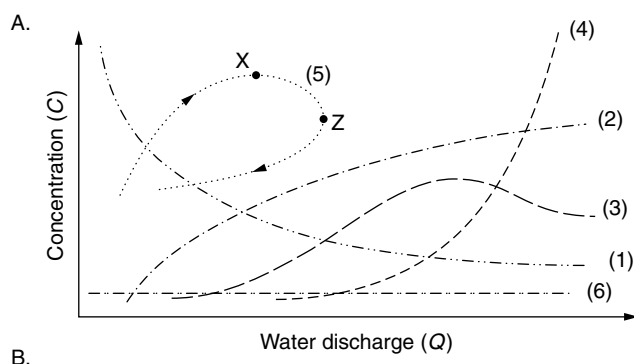


Figure 3. Patterns of concentration from discharges into streams/rivers.

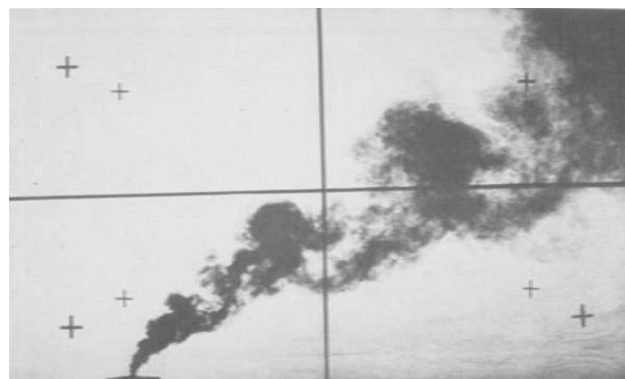


Figure 4. A single port buoyant jet in cross flow deflected by the ambient current (near-field flow).

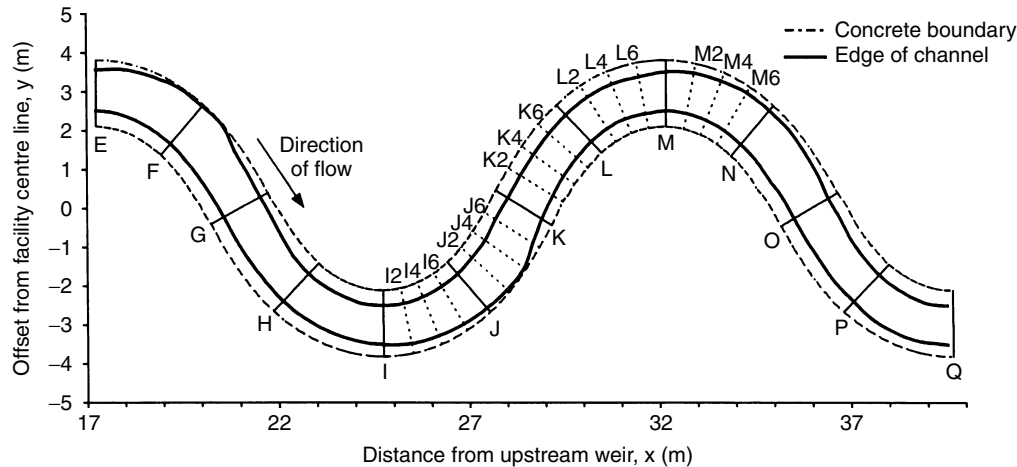


Figure 5. Typical channel geometry and section numbering.

The distance below the outfall where the effluent spreads across the channel width is termed “crossing distance.” Within the mixing zone, apportion of the cross section termed “limited use zone” (LUZ) or “zone of noncompliance” occurs wherein the concentration of a pollutant may not comply with a specified water quality objective. The remaining portion of the cross section known as the “zone of passage” serves as a suitable habitat for fish and other desirable aquatic life. Note that in the far-field region, the mixing of effluent with the river water takes place due to lateral and longitudinal dispersion. A typical example of meandering given by Boxall et al. (3) is shown in Fig. 5. The cross-sectional profile and velocity pattern of flow at different sections of the channel are shown in Fig. 6. It can be seen from that figure that the maximum velocity profiles are always on the convex (right) portion of the channel looking downstream. When a pollutant is discharged at the bottom or on the surface of the different sections, the length of the mixing zone varies due to variation in velocity profiles. Also, the location of injection of pollutants (on the surface or at the bottom) changes the mixing phenomena, which can be well observed in Fig. 7.

It can be seen from the Fig. 7 that vertical mixing is usually complete in a short distance below the outfall or injection point, whereas lateral spreading is more gradual resulting in lateral concentration gradients. Ultimately, the cross-sectional concentration distribution attains uniformity at some distance below the outfall.

Mixing zones are prohibited in ephemeral waters or where there is no water for dilution (4). In a mixing zone, it is proposed to have high in-stream discharge for proper dilution and to meet standards at a boundary for the balance of the waterbody. It is the means for expediting mixing and dispersion of sewage, industrial waste, or other waste effluents in receiving waters.

Early workers (5–15) in the field devised mixing-zone concepts based on the lateral, vertical, and longitudinal dispersion characteristics of the receiving waters. Formulas predicting space and time requirements for diluting certain pollutants to preselected concentrations were also developed. The mixing behavior of a discharge depends largely on the depth of the ambient water, the momentum

and buoyancy of the discharge, the spatial orientation of the discharge system, and the effects of many other factors. As a result, the mixing zone and mixing length are essential variables required for assessing the pollutant concentration/load downstream of the mixing zone.

Consideration of how to compute the distance from an outfall to compute mixing is a separate more complicated topic. However, the order of magnitude of the distance from a single point source to the zone of complete mixing is obtained from (16)

$$L_m = 2.6U \frac{B^2}{H} \quad (2)$$

for a side bank discharge, and from

$$L_m = 1.3U \frac{B^2}{H} \quad (3)$$

for a midstream discharge. In these equations,

L_m = distance from the source to the zone where the discharge has been well mixed laterally, in ft,

U = average stream velocity in fps,

B = average stream width in ft,

H = average stream depth in ft

DESIGN FLOW

The central problem in water quality management is assigning allowable discharges to a waterbody so that a designated water use and quality standard is met using basic cost–benefit principles. The following steps are required for proper design flow and waste load allocation in streams/rivers:

- evaluation of waste load input and in-stream discharge
- dilution phenomena and water quality response
- mixing length of pollutants in streams/rivers

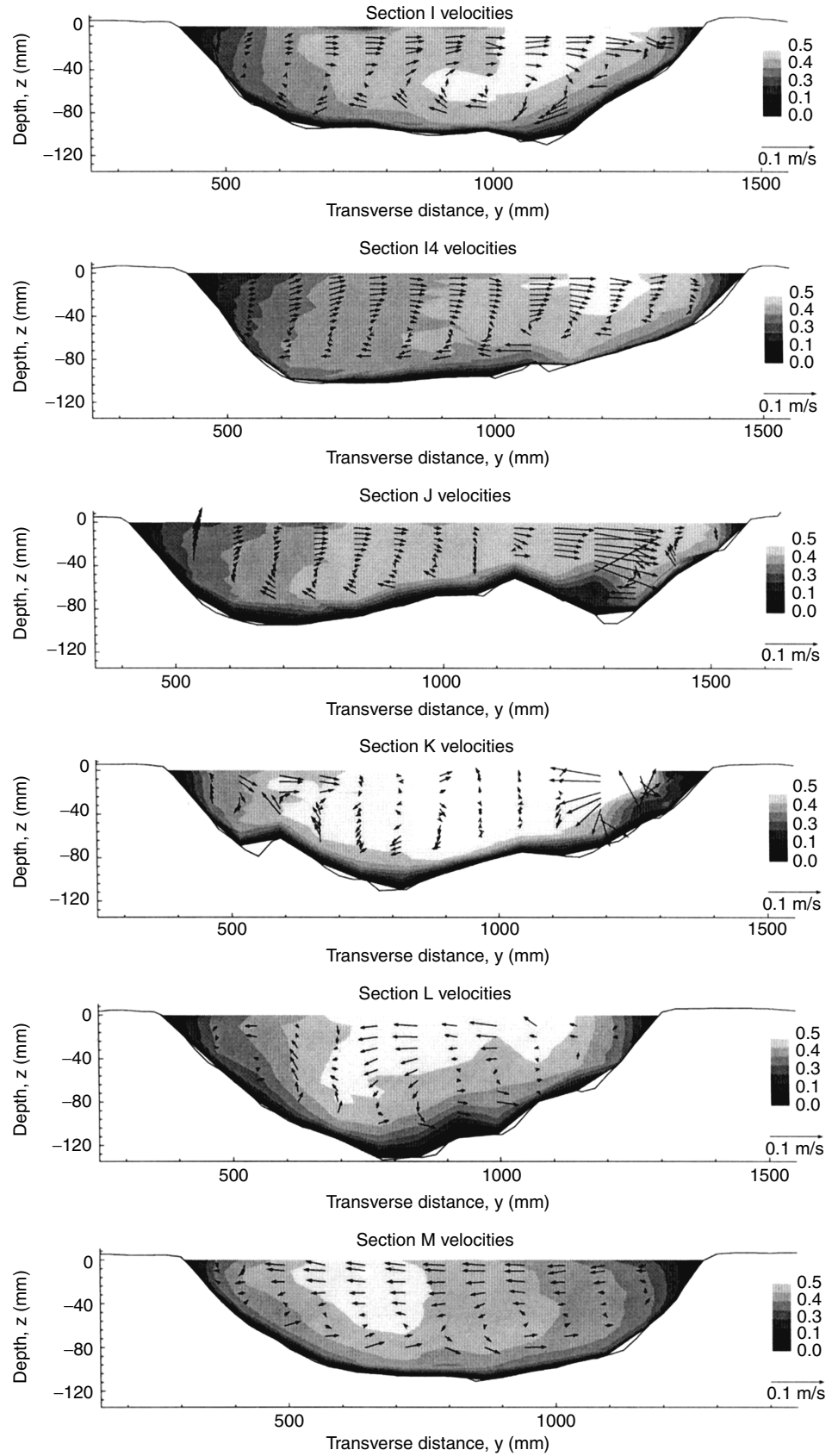


Figure 6. Cross-sectional and velocity profiles at different sections.

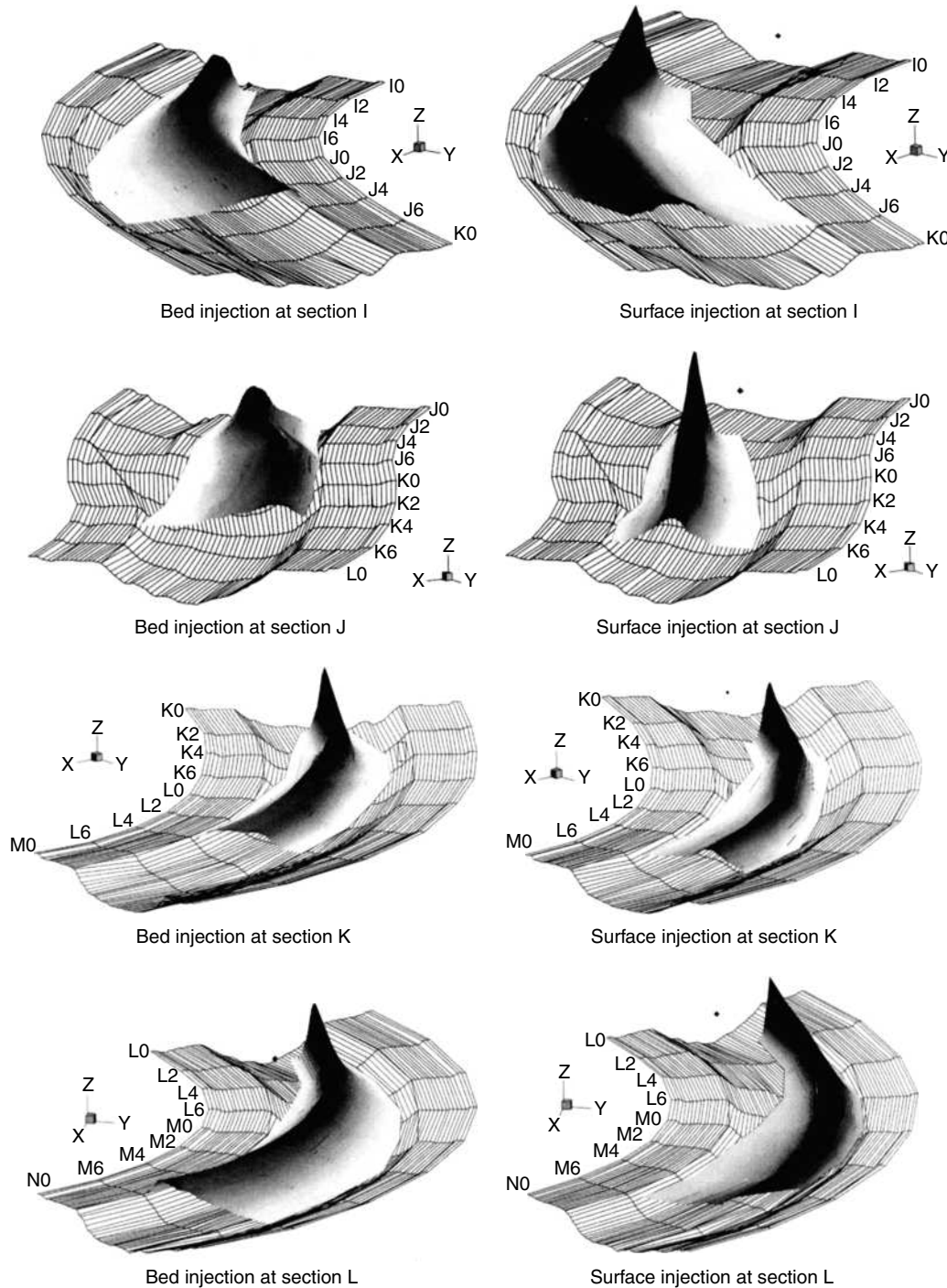


Figure 7. Three-dimensional representation of dye tracer measurements below different injection points.

- desirable water use and evaluation of water quality criteria that will permit such uses
- relationship between load and water quality and selection of projected conditions
- It is generally not sufficient to make a scientific engineering analysis of the effect of waste load inputs on water quality. The analytical framework should also include economic impacts which, in turn, must

also recognize the sociopolitical constraints that are operative in the overall problem context.

To illustrate overall waste load allocation and a design flow problem for dissolved oxygen, a schematic diagram given by Thomman and Muller (17) is presented (Fig. 8).

It can be seen from Fig. 8 that for properly allocated waste loads in streams/rivers, DO values do not fall below the standard limits (as shown in the lower part of the figure). In the figure, there are several points at which

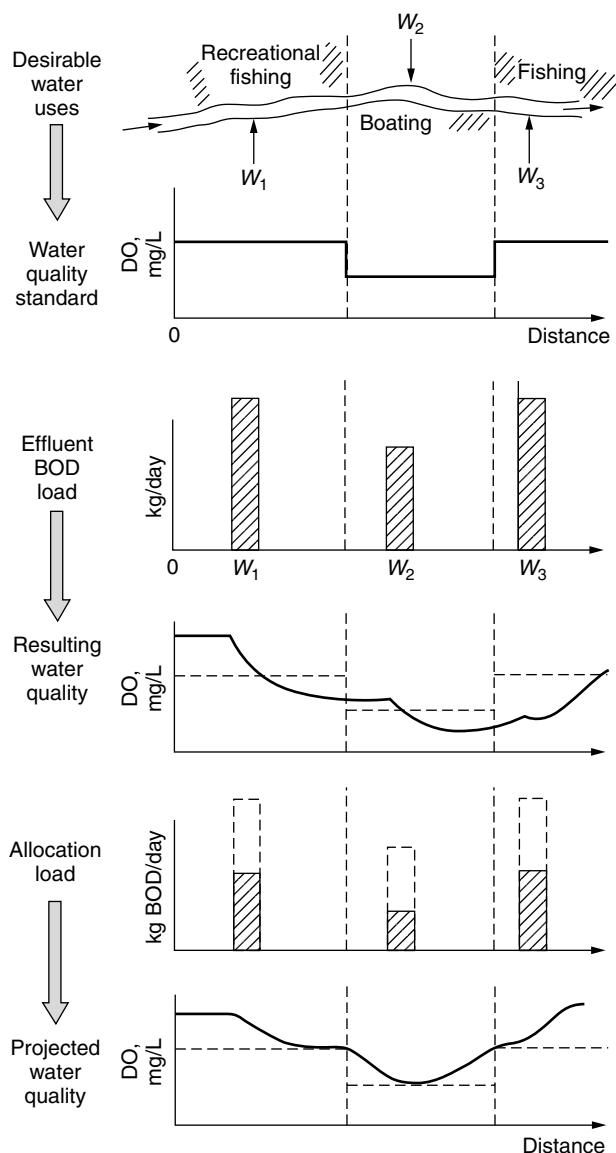


Figure 8. Representation of waste load allocation problem for dissolved oxygen.

careful judgments are required to provide a permissible waste load allocation. Determination of design conditions, including flow and parameters, must be evaluated for a waste load allocation. The specification or projection of flow and parameter conditions under a given design is a most critical step and is a combination of engineering judgment and sensitivity. The concentration at the outfall can be controlled by

- (1) reducing the effluent concentration of the waste input by (a) wastewater treatment, (b) industrial in-plant process control, and (c) eliminating effluent constituents by pretreatment prior to discharge to the municipal sewer or by different product manufacturing for an industry;
- (2) reducing the upstream concentration by upstream point and nonpoint source controls;

- (3) reducing the effluent volume by (a) reduction in infiltration into the municipal sewer system, (b) reduction of direct industrial discharge volumes into the sewer system, and (c) reduction, for industry, of waste volume through process modifications;
- (4) increasing the upstream flow by low flow augmentation, that is, release from upstream reservoir storage or from diversions from nearby bodies of water;
- (5) increasing the environmental, in-stream degradation rate of the substance.

TYPICAL EXAMPLES OF DILUTION IN STREAMS/RIVERS OF INDIA

In India, all 14 major rivers including Ganga, Yamuna, Gomti, Cauvery, and Damodar are polluted. The Rivers Damodar and Yamuna are the most heavily polluted. Many parts of these rivers do not have any dissolved oxygen, and no wonder that they fail to support the growth of desirable aquatic fauna and flora. The River Ganges is heavily polluted at Allahabad, Kanpur, Varanasi, Patna, and Bhagalpur. The cross section and BOD/DO contours at Kanpur and Varanasi are shown in Figs. 9 and 10, respectively (18).

As can be seen from Figs. 9 and 10, the BOD concentrations found are very high at the pollutant outfall site. There is a significant amount of variation in BOD across the river. A similar situation has been observed in the longitudinal direction also due to dilution and mixing. The variation in dissolved oxygen (DO) downstream of the sewage outfall on the River Ganges, at Varanasi is shown in Fig. 11. As can be seen, the DO pattern at Assi Ghat in Varanasi, India, was monitored at different locations in longitudinal and transverse directions (19,20), and the river shows a nearly perfect dilution ratio. Similar studies were also, carried out for River Kali, a highly polluted river in India (18,21).

SUMMARY

Dilution, mixing zone, and design flows are interlinked issues that have been addressed here. Now, a better understanding of the mixing zone is available in the literature, and there are softwares like CORMIX (MixZon Inc., 2003) that can very elegantly perform mixing zone analysis. In the literature, near-field and far-field analysis of mixing of pollutants has also received attention. However, there are still challenges ahead, and these are related to the presence of a dead zone, adverse flow gradients, and curvilinear flows.

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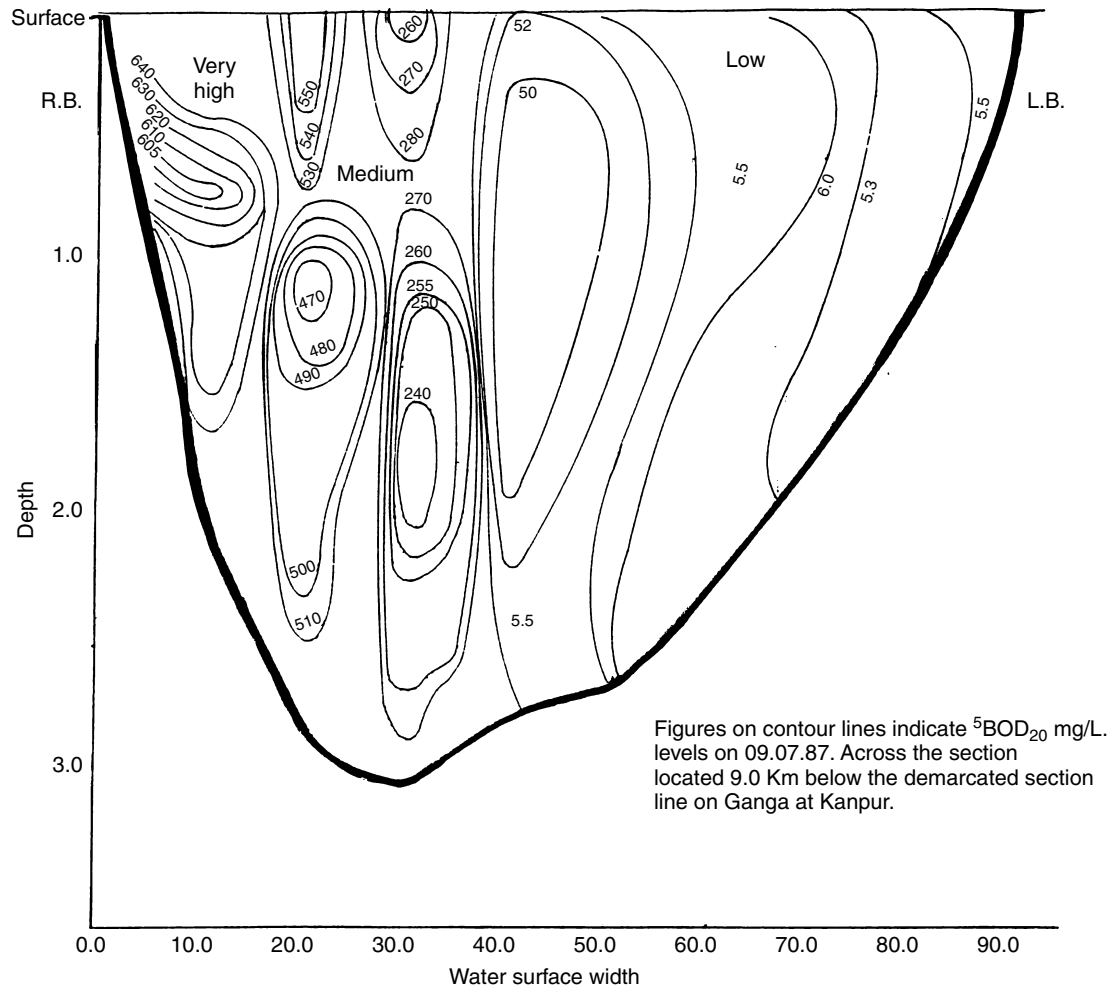


Figure 9. Cross sectional profile of BOD levels in the Ganga at Kanpur.

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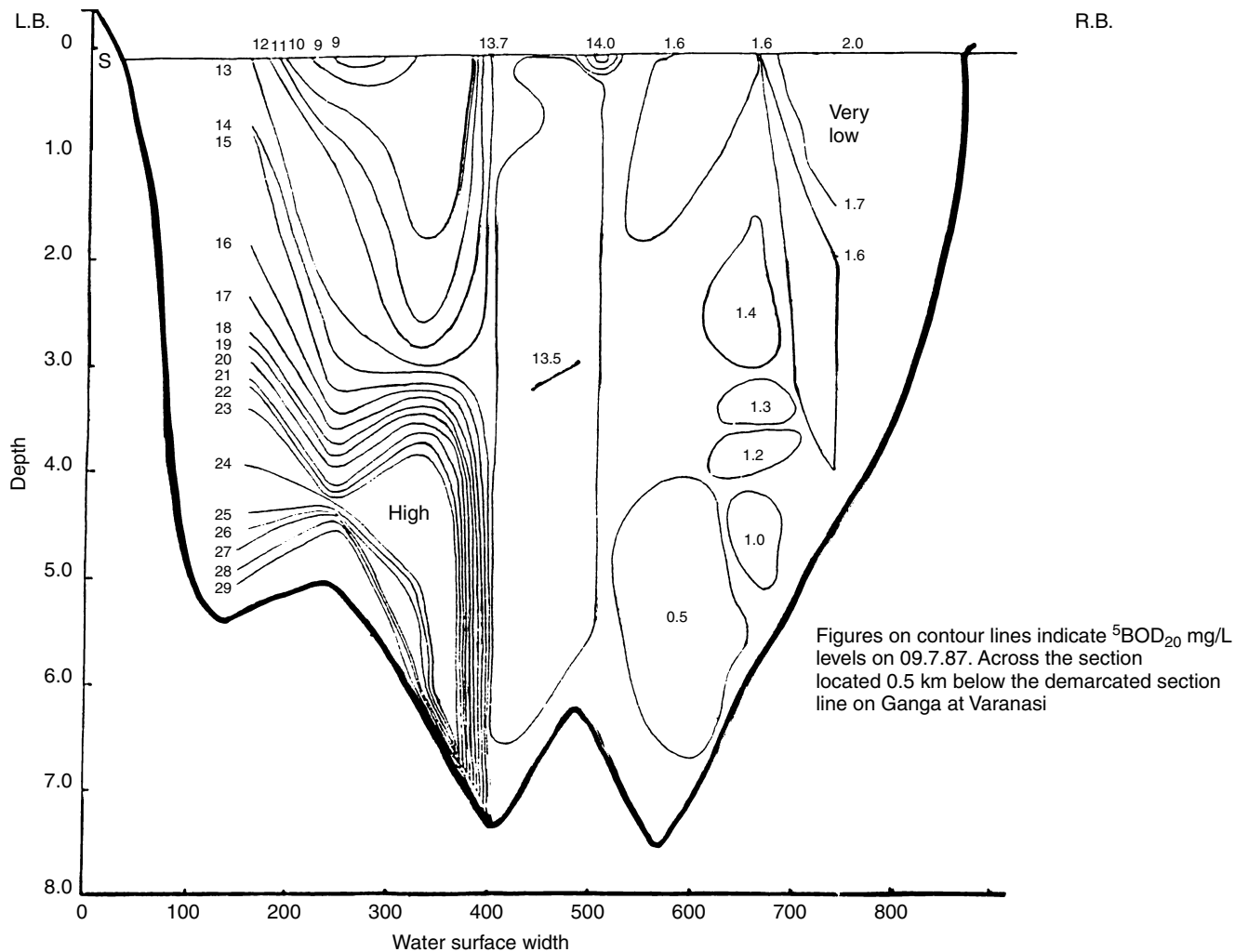


Figure 10. Cross-sectional profile of BOD levels in the Ganga at Kanpur.

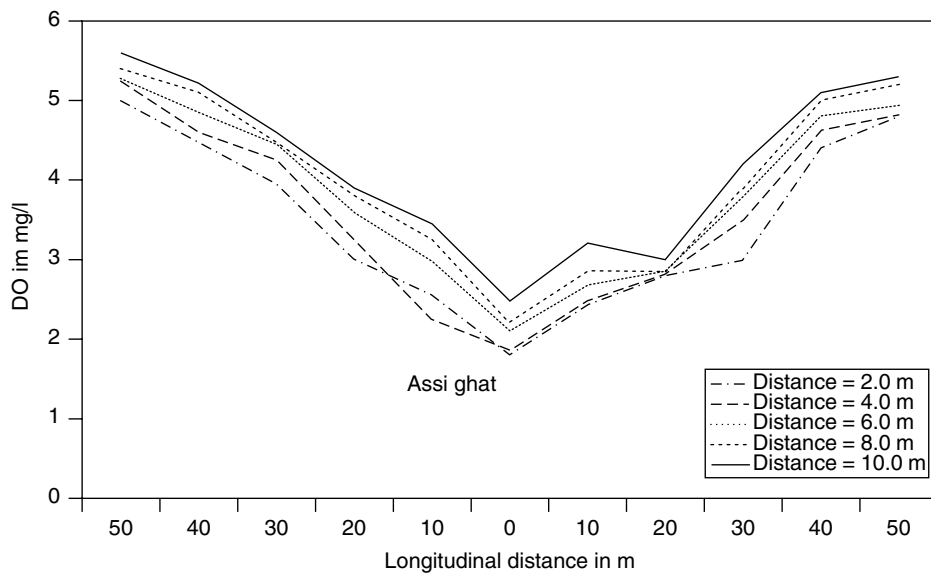


Figure 11. DO variations in the River Ganges at Assighat, Varanasi, India.

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DRAINAGE DITCHES

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OVERVIEW

Drainage ditches are common landscape features found in urban, industrial, and agricultural settings. Just as diverse as the environment in which they can be found, ditches have a wide array of physical and structural parameters. They may be lined by concrete, rock (rip rap), bare soil, or vegetation. Some ditches are small enough to step across, while others may require different methods of moving from one side to the opposite. Ditches, by their very nature, are water conveyance features. Natural ditches, formed as gullies, have been prominent landscape features in areas of topographical relief since the Earth was formed. Investigations of historical gully formation find ditches associated with primitive slash-and-burn agriculture. Drainage ditches have historically been efficient methods for transporting water away from moist or wet areas. The most obvious illustration is found in agricultural fields, where production acreage is surrounded by a myriad of drainage ditches (Figs. 1–3). Moving stormwater or excess irrigation water away from crops is the primary focus of agricultural drainage ditches.

In industrial areas, ditches are often the first postmanufacturing “treatment” area for discharges before the effluent reaches a receiving stream. The discharge pipe often empties into a vegetated drainage ditch, which will transport water to a river, lake, or stream. Urban settings are laced with interspersed ditches of varying sizes, again with the primary objective of removing stormwater from residential areas. Many urban ditches are lined with concrete, which further encourages water movement. Unfortunately, this also negates many positive



Figure 1. Constructed ditch draining farm field.



Figure 2. Overgrown constructed ditch draining farm field.



Figure 3. Natural drainage canal in farm field.

water quality enhancement opportunities that drainage ditches possess.

Ditches may also serve to deliver water to areas for irrigation purposes. In western states where water supplies are closely managed, ditches provide an important conduit for water transportation. The California Gold Rush resulted in construction of over 5000 miles of canals to

transport water to gold fields. Today, California supplies over half of the fruits, vegetables, and nuts for the United States (plus exports) on 3% of the nation's cropland because of its 9.5 million irrigated acres, due in part to the extensive network of canals and ditches. Thus, the topic of drainage ditches covers a broad spectrum of conveyance systems in time and space located from agricultural to urban settings. For our purposes, discussion is focused on ditches located in agricultural settings.

Engineering techniques associated with ditches have evolved so that an excellent level of confidence can be achieved in ditch design. Studies of open channel flow began in natural channels, but major knowledge breakthroughs have come through investigations of constructed channels. The use of laboratory flumes in the twentieth century greatly accelerated the knowledge base necessary for the current area of challenge—modeling. Energy, friction, channel roughness, Froude numbers, and critical depth of flow are some of the parameters where historical research provides the basis for aquatic habitat design.

Although drainage works have been in existence for over 2500 years, they are not without controversy (1). During the Middle Ages, ditches exsiccated English fens to increase agricultural production landscape. While beneficial for farmers, fen draining was seen as a direct threat to both fishermen and fowlers (1). In the early to mid-twentieth century, the U.S. government strongly favored drainage of wetlands and swamps to boost agricultural

acreage, facilitate transportation, and combat mosquito and other vector-borne illnesses (2). As a result, fewer than half of the original U.S. wetlands remain (1). Without drainage, however, the fertile Midwest states of Ohio, Indiana, Illinois, and Iowa would still be largely swamps or land too wet for viable agriculture production (1). The last congressionally mandated decennial census of U.S. drainage (1985) indicated 110 million acres of agricultural land profited from drainage, and the net capital value of farm drainage work was estimated at \$25 billion (1).

The U.S. Department of Agriculture's Natural Resource Conservation Service (USDA NRCS) promotes the use of conservation practices to improve soil and water quality. Under the broad topic of conservation are two standards for excess water removal, Practice 607 (Surface Drainage, Field Ditch) and Practice 608 (Surface Drainage, Main or Lateral) (3). Neither specifically addresses water quality enhancement.

Data in Tables 1–5 summarize available global drainage ditch research from the 1960s to 2004. As is common in progression of research, initial studies searched for basic information (e.g., surveys or nutrient loading). Later research focused more on current themes.

CURRENT AND FUTURE DIRECTIONS

Currently, experimental agricultural drainage ditch research is being primarily conducted with three institutions in the United States: Arkansas State University

Table 1. Ecological Research Conducted on Agricultural Drainage Ditches

Research	Location	Results	Reference
Floristic survey	Eastern Saudi Arabia	More than 100 plant species in canals	4
Flora	The Netherlands	78% of <i>Spirogyra</i> (algae) in Netherlands found in ditches and pools; some ditches had >20 species	5
Flora	United States	Maintenance of weeds on ditch banks	6–8
Flora	Mississippi Delta, United States	Ditches without surrounding riparian cover have increased nutrient concentrations	9
Macroinvertebrate	Rhône Delta, France	Description of copepod, <i>Cletocampus retrogressus</i>	10
Macroinvertebrate	Germany	Discovery of aquatic beetle, <i>Dytiscus semisulcatus</i>	11
Macroinvertebrate	United Kingdom	Ephemeroptera and Plecoptera present in nutrient-enriched ditches	12
Macroinvertebrate	Wicken Fen, United Kingdom	Rotation of ditch maintenance allowed Ephemeroptera establishment and survival	13
Macroinvertebrate	United Kingdom	Importance of vegetated refuges to survival of uncommon gastropods	14,15
Macroinvertebrate	Yakima Valley, Washington, United States	Successful overwintering of beneficial and harmful insects in ditches	16
Fish	Germany	Suggested management practices to offset fish population damages	17
Macrofauna	The Netherlands	Random distribution of macrofauna taxa in ditch sampling analyses	18
Fauna	Canada	90% of fish species utilizing drainage ditch habitat spawned in ditch	19
Fauna	United Kingdom	Ditch age and bank profile contributed to differences in individual ditch fauna	20
Flora/fauna	Elbe Estuary, Germany	Floral/faunal shifts as a result of pesticide applications and eutrophication in ditches	21
Plant and invertebrate interaction	Germany	<i>Anopheles</i> (mosquito) occurrence related to ditch plant succession	22
Floodplain diversity	Lower Frome floodplain, United Kingdom	Slow moving Rushton Ditch contributed largely to floodplain biodiversity	23

Table 2. General Research Conducted on Agricultural Drainage Ditches

Research	Location	Results	Reference
Design criteria	Malaysia	Integrate engineering and agricultural practices to maximize land productivity	24
Drain cleaning	Australia	Necessity of cleared drains in sugarcane production to maintain ditch carrying capacity	25
Maintenance	United Kingdom	Recommend dredging cycle of 3 years to maintain diverse floating and aquatic plants	26
Mechanical harvesting	The Netherlands	Determination of plant cutting times for maximum species richness on sand and peat soils	27
Upland ditches	United Kingdom	Little ecological advantage gained with ditches (grips) in uplands	28
Model	The Netherlands	Modeling determined critical nutrient loading dependence on ditch depth	29,30
Watershed hydrology	France	Differences seen between watersheds with and without ditches	31
Hydrology	United Kingdom	Physical-based hill slope model to understand changing catchment runoff regimes	32
Hydrology	United Kingdom	Drain interaction with channel hydrology (DITCH) model indicated higher ditch water levels did not improve ecology	33
Hydrology	United States	Correspondence between peak flows of ditches and rivers	34
Best management practice (BMP)	United States	Suggested use of ditches as BMP for nutrient and pesticide mitigation	35

Table 3. Nutrient Research Conducted on Agricultural Drainage Ditches

Research	Location	Results	Reference
Phosphorus	Australia	Decrease in phosphorus after traveling through bare drains; increase in phosphorus after passing through grass pasture	36
Phosphorus	New Zealand	Drainage sediments act as phosphorus sinks	37
Phosphorus	Delaware, United States	Ditch sediment biologically reactive phosphorus similar to that of topsoils of adjacent fields	38,39
Nitrate	River Eider Valley, Germany	Increase in retention time resulted in decrease in nitrate due to vegetation and organic debris accumulation	40
Nutrients	The Netherlands	No increase in phytoplankton in nutrient-enriched ditches	41
Nutrients	The Netherlands	90–95% phosphorus removal; low nitrogen removal	42
Nutrients	St. Lucie County, Florida, United States	Nutrient concentrations in citrus grove ditch water higher than concentrations in vegetable farm ditch water	43
Nutrients	United States	Ditches around baitfish farms are of benefit	44
Nutrients	Arkansas and Mississippi Deltas, United States	Ditches help remediate aquaculture effluent	45
Nutrient loading	United States	Seasonal variation of nutrients by distinctive ditch size	46,47
Nutrient standards	The Netherlands	Failure of general standards to support minimum water; need to be specific	48
Nutrients + pesticides	The Netherlands	Increase in macrophyte biomass and stored nitrogen and phosphorus	49
Herbicide + nutrient loading	South Saskatchewan River, Canada	Loadings diluted by storm; no significant increase in concentration	50

(ASU), Ohio State University (OSU), and the University of Minnesota (UM). ASU's research site includes a series of newly constructed vegetated drainage ditches, catchment ponds, small meanders, and riparian areas. Innovative research, in conjunction with the USDA Agricultural Research Service's National Sedimentation Laboratory, will highlight contaminant fate and transport, partitioning, mitigation, and ecological benefits of ditches alongside agronomic research. OSU continues to conduct research on the benefits of one-stage and two-stage ditches for longevity, water quality, and ecological enhancements (73). The University of Minnesota Southwest Research and Outreach Center recently constructed

two 200-m open drainage ditches to study sediment and nutrient removal from drainage water (74).

Our literature review indicated ditch research is centered in the United States, The Netherlands, the United Kingdom, and Germany. Research thrusts range across topics of this decade, including greater understanding of basic hydrological and trapping processes, ditch contributions to pollution, contaminant processing by ditches, and maintenance. Future investigations must view drainage ditches holistically as water conveyance segments of a watershed. While basic studies on habitat, maintenance, and function are needed, research must define the ditch's role as a key facet of edge-of-field conservation

Table 4. Pesticide Research Conducted on Agricultural Drainage Ditches

Research	Location	Results	Reference
Herbicide retention	France	70% decrease in diflufenican with ditch surface contact	51
Herbicide transport	United Kingdom	99% of sulfosulfuron load found in first 4% of flow	52
Herbicide transport	United Kingdom	Mecoprop transport following storm event exceeded U.K. concentration guidelines	53
Herbicide (Linuron)	The Netherlands	Amendment decreased pH and dissolved oxygen; no effect on plant composition; diatoms and cryptophytes decreased; cladocera and copepods increased	54–56
Insecticide effects	The Netherlands	Vegetated systems (<i>Elodea</i>) sorbed the majority of applied chlorpyrifos; sediment was sink where no vegetation present; effects on macroinvertebrates occurred four times faster in open versus vegetated water	57–59
Lambda-cyhalothrin (insecticide)	The Netherlands	Sediment–pesticide sorption lower in areas with vegetation; alkaline hydrolysis main pesticide transformation process	60
Esfenvalerate	Mississippi Delta, United States	Toxicity measured in ditch; concentrations from 50-acre runoff mitigated to no observed effects levels in 510 m of vegetated ditch	61,62
Atrazine + lambda-cyhalothrin	Mississippi Delta, United States	Reduce pesticides from 5-acre runoff to no observed effects level in 50 m of vegetated ditch	63
Bifenthrin + lambda-cyhalothrin	Mississippi Delta, United States	Reduce pesticides from 50-acre runoff to no observed effects level in 650 m of vegetated ditch	64,65
Pesticides	California, United States	78% of ditch samples toxic—affected estuary	66
Spray drift buffer	The Netherlands	Buffer zone (3-m width) around ditch banks lowered spray drift by 95% in ditches	67
Insecticide + nutrients	The Netherlands	Increase in phytoplankton abundance; grazers effected by pesticide amendment	41
Insecticide + nutrients	The Netherlands	Increase in macrophyte biomass and stored nitrogen and phosphorus	49
Herbicide + nutrient loading	South Saskatchewan River, Canada	Loadings diluted by storm; no significant increase in concentration	50
Monitoring	Lower Fraser Valley, Canada	Consistent presence of diazinon and dimethoate in ditch water	68
Monitoring	Canada	4% of sampled ditches were toxic to <i>Ceriodaphnia dubia</i> ; 14% impaired <i>C. dubia</i> reproduction; trace amounts of organophosphate insecticides	69

Table 5. Sediment Research Conducted on Agricultural Drainage Ditches

Research	Location	Results	Reference
Sediment	United States	Danger of Mississippi and Illinois River agricultural field ditches eroding	70
Sediment	United States	Cost is \$0.45 per cropland acre to remove Ohio sediment from ditches; losses could be reduced by 25% with best management practices	71
Sediment	United States	Decreased sediment removal cost by implementing best management practices in Illinois, Indiana, Ohio, and Idaho	72

and pollution reduction systems. Given negative aspects of direct drainage, modifications that ameliorate ditches should blend habitat improvement with utilitarian design.

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DRAINAGE NETWORKS

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A drainage network is the configuration of the stream courses in a river basin. Drainage networks are the products of fluvial erosion processes. As erosion incises at a point on the land surface, that point becomes lower, which allows it to collect more flow. As the flow increases, the erosion rate also increases. This positive feedback ultimately produces a dendritic network of channels within a river basin. The drainage network is responsible for the surface transport of water, sediment, and contaminants out of the basin.

EXTRACTION OF DRAINAGE NETWORKS FROM DIGITAL DATA

Channel networks are commonly extracted from digital elevation models (DEMs), which are considered alternatives to traditional topographic maps. Before the drainage net is extracted, the DEM is usually processed to remove pits and flat areas that are artificially generated in the production of the DEM. To define the channel network, the direction that water flows on the surface must be determined for every grid point in the DEM. Then, the amount of flow passing through each point can be calculated and the channel heads can be identified.

The first task is the identification of the flow directions. The three prevailing methods are (1) single flow directions, (2) multiple flow directions, and (3) continuous flow directions. When single flow directions are used, water is assumed to travel from a grid point to the neighbor that produces the steepest downward slope (1). When multiple flow directions are used, the flow from a grid point is distributed to all neighbors that are lower than the grid point. The proportion of flow that each neighbor receives depends on the slope produced between the grid point and that neighbor (2). The method of single flow directions is usually adequate for convergent parts of the topography (e.g., valley bottoms), but multiple flow directions is more accurate for areas of divergent flow (e.g., hilltops). The method of continuous flow directions assigns a flow direction in any direction, not just the directions of the eight neighbors (3). Eight triangular facets are formed by connecting a grid point with its neighboring points, and the downslope vector is determined for each of these facets. The flow direction associated with the grid point is the direction of the steepest downslope vector from all eight adjoining facets. Once the flow directions are determined from any of these methods, the number of grid cells (or topographic area) whose flow would pass through a selected grid point can be determined. This accumulated area is known as the contributing area or drainage area.

One key issue in deriving channel networks from DEMs is the identification of channel heads. Flow directions and contributing areas are defined for all grid cells

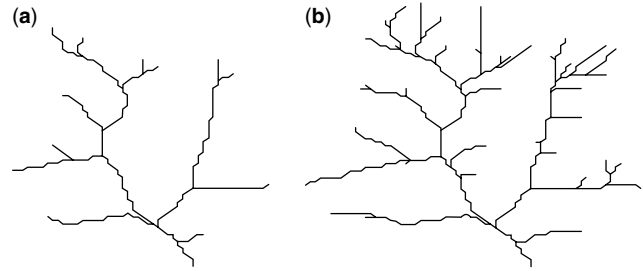


Figure 1. The effect of different critical contributing areas on drainage network extraction from a DEM: (a) with a threshold of 125 pixels and (b) with a threshold of 25 pixels.

in a DEM, but not all grid cells contain a channel. Channel heads are widely defined as the points where fluvial processes become important agents of erosion. The simplest approach for channel head identification is to select a contributing area threshold (4). One way to select this threshold is to identify the contributing area at which “feathering” of the channel network occurs. When the threshold is too small, the smallest channels will often run parallel to each other making the network appear feathered. Figure 1 shows the effect of choosing different contributing area thresholds on the drainage net extracted from a DEM. A series of field studies (5) found that the location of channel heads is better described as a threshold that depends on both the contributing area and the local slope. Observations also suggest that real channel networks are usually much more extensive than the blue lines on USGS quadrangle maps.

STATISTICAL PROPERTIES OF DRAINAGE NETWORKS

The Strahler ordering system (6) is the most common method to identify distinct channels within a channel network. The rules of this ordering system are (1) streams beginning at a channel head are order 1; (2) when two streams of equal order ω meet, a stream of order $\omega + 1$ is created; and (3) when streams of unequal order meet (e.g., ω and $\omega - 1$), the channel segment immediately downstream has the higher order of the two joining streams (e.g., ω). Figure 2 shows an example of the Strahler ordering system. The largest stream order in a basin is common as a qualitative measure of basin size.

The relationship between streams of different order can be described by Horton’s laws (7), which include the laws of stream numbers, stream lengths, and basin areas. The law of stream numbers is expressed as

$$\frac{N(\omega)}{N(\omega + 1)} = R_B$$

where $N(\omega)$ is the number of streams of order ω and R_B is called the bifurcation ratio. The bifurcation ratio characterizes the propensity of the drainage network to branch. The law of stream lengths is expressed as

$$\frac{\bar{L}(\omega + 1)}{\bar{L}(\omega)} = R_L$$

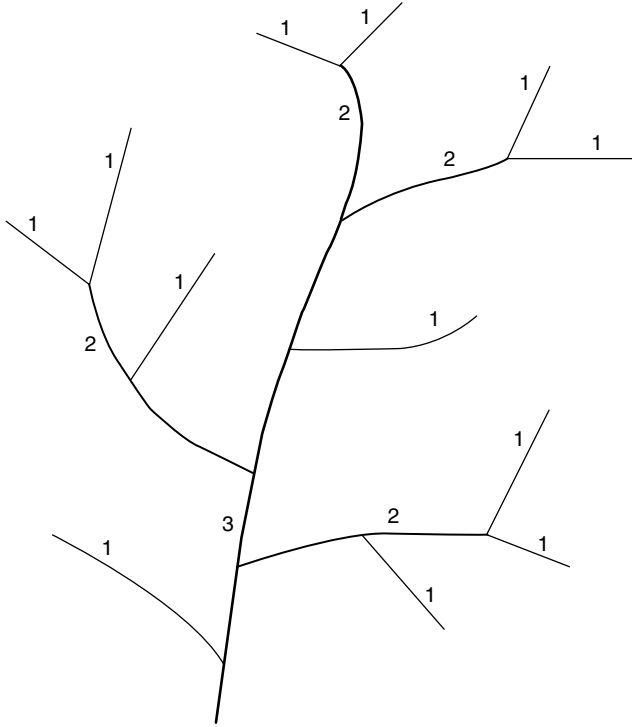


Figure 2. Strahler ordering of a third-order river basin.

where $\bar{L}(\omega)$ is the mean length of streams of order ω and R_L is called the length ratio. The length ratio describes the typical increase in stream length that occurs with increasing stream order. The law of basin areas is expressed as

$$\frac{\bar{A}(\omega + 1)}{\bar{A}(\omega)} = R_A$$

where $\bar{A}(\omega)$ is the mean area contributing to streams of order ω and R_A is called the area ratio. Typical values for R_B , R_L , and R_A are 4, 2, and 5, respectively. The fact that Horton's laws hold for most real and simulated drainage networks has led some researchers to suggest that the laws are largely a product of the ordering system and thus not particularly useful for differentiating between different networks.

Natural drainage networks are also considered to be scaling-invariant or fractal objects. Loosely speaking, a fractal is an object that appears statistically similar when viewed up-close at a fine resolution or from a distance at a coarse resolution. Several empirical characteristics of river basins are products of this property. For example, Hack's law (8) describes the power-law relation between the length of a basin's main channel L and the basin's area A :

$$L \propto A^{0.57}$$

The main channel can be defined by identifying the longest channel in a basin when measured from the basin outlet to the channel head along the stream. The fact that the exponent in Hack's law is above 1/2 (the value one would expect from simple geometry) has been shown to

be a result of the fractal sinuosity of the stream course. Another product of the scaling-invariance is the slope-area law, which can be written:

$$\bar{S} \propto A^{-\theta}$$

where A is contributing area (or equivalently basin area) and θ is a scaling exponent, which typically ranges from 0.3 to 0.7. $\bar{S}$ is the mean local slope for channel locations that drain basins with area A .

Some impacts of channel networks on the hydrologic response of a basin can be characterized by the drainage density and width function. The drainage density D_d is a measure of topographic texture and stream spacing and can be defined as

$$D_d = \frac{L_T}{A}$$

where L_T is the total length of stream channels in a basin. The drainage density is closely related to the average hillslope length and has been widely employed to predict characteristics of runoff production and the hydrologic response to precipitation events.

The width function is a graph that describes the probability distribution of flow path lengths in a basin. The x -axis of a width function spans the range of distances between the grid points in a basin and the basin outlet, where distance is measured along the flow paths. The width function's y -coordinate indicates the number of grid points that have the specified distance from the basin outlet (topographic area is often used instead of the number of grid points). Notice that the width function is closely related to the basin's unit hydrograph. In fact, if flow speed was constant throughout a basin, the width function would be nearly equivalent to the unit hydrograph.

The basin circularity ratio R_C (9) is a measure of the basin shape and can be defined as

$$R_C = \frac{4\pi A}{P^2}$$

where P is the length of the basin perimeter. It indicates how closely the basin resembles a circle. The closer the circularity is to 1, the greater the similarity to a circle. Another measure of basin shape is the form factor R_F , which can be written as

$$R_F = \frac{A}{L_x^2}$$

where L_x is the maximum length between the basin outlet and the opposite boundary (10). Numerous other measures of basin shape are available. More elongated basins tend to have hydrologic responses that are more distributed in time, whereas compact basins tend to exhibit more peaked hydrographs.

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DYES AS HYDROLOGICAL TRACERS

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INTRODUCTION

Dye tracers have been used in hydrological investigations for more than a century. In 1877, uranine (fluorescein) was used as a tracer to test the hydraulic connection between the Danube River and the Ach Spring in southern Germany (1). In 1883, the French physician des Carrières successfully proved the source of a typhus epidemic in the city of Auxerre by conducting a tracer experiment with the dye aniline (2). First systematic investigations on the suitability of dyes as tracers were conducted even before the turn of the century (3). Subsequently, the use of dyes as tracers became a common practice in hydrological investigations (4–6).

A classic example of the use of dyes in hydrology is the study of residence times and pathway connectivities in Karst (7). Further applications range from studying dispersion in streams and lakes, to determine sources of water pollution, and to evaluate sewage systems. In the vadose zone, dyes have been mainly used to visualize flow patterns (6).

Thousands of different dyes are commercially available (8), but only a few are suitable for hydrological investigations. Many dyes have been studied specifically for their suitability as hydrological tracers, and recommendations

were made on “best” dye tracers (9–12). Depending on the specific applications, different chemical characteristics of a dye may be desirable. For instance, for visualization of water flow in soils, a dye should be clearly visible and trace the water movement accurately. In this particular case, the dye will preferably be blue, red, green, or fluorescent to contrast distinctly from the soil background. The accurate tracing of the water movement demands that the dye does not sorb too strongly to subsurface materials, which poses limitations on the chemical characteristics of a dye.

Here, we summarize tracer characteristics and the applications of dye tracers in surface and subsurface hydrology. We then discuss the limitations and potential problems in using dyes for tracing water flow and solute movement. Selection of an appropriate dye is critical for the success of a tracing study. We present a case study on the application of quantitative structure-activity relationships (QSARs) for screening, selecting, and designing optimal dye tracers for a specific use.

TRACER CHARACTERISTICS OF DYES

Dye tracers, particularly fluorescent dyes, are often preferred over several other types of tracing materials because of their unique characteristics. Many dyes (1) can be readily detected at a concentration as low as a few micrograms per liter, (2) can be quantified with simple and readily available analytical equipment, (3) are nontoxic at low concentrations, and (4) are inexpensive and commercially available in large quantity (7,12,13). In addition, because of their coloring properties, dyes allow us to visualize flow pathways in the subsurface. Many dye tracing studies conducted in the past 10 years have clearly demonstrated that flow patterns in the subsurface are often highly irregular; an example of a nonuniform infiltration front in a sandy soil is shown in Fig. 1.

Despite these desirable characteristics, important drawbacks exist in using dye tracers. Dye tracers are not conservative tracers; i.e., they sorb to subsurface media and do not necessarily move at the same speed as the water to be traced. The sorption behavior of dyes is influenced by the properties of the subsurface materials and the chemistry of the aqueous phase (9,12,14–16). Some dyes degrade when they are exposed to sunlight, e.g., uranine (10,17,18), and some can be degraded by microorganisms. Consequently, dye tracers may behave differently under different natural environments. Thus, the suitability of dye tracers should be tested before they are used in hydrological studies.

An “ideal” water tracer is a substance that (1) has conservative behavior (i.e., does not sorb to solid media, is resistant to degradation, and stable in different chemical environments); (2) does not occur naturally in high concentrations in the system to be investigated; (3) is inexpensive, (4) is easy to apply, sample, and analyze; and (e) is nontoxic to humans, animals, and plants (6). These requirements are difficult to meet for a single chemical. Different types of dyes have been proposed as best suitable water tracers, and these dyes are discussed below.

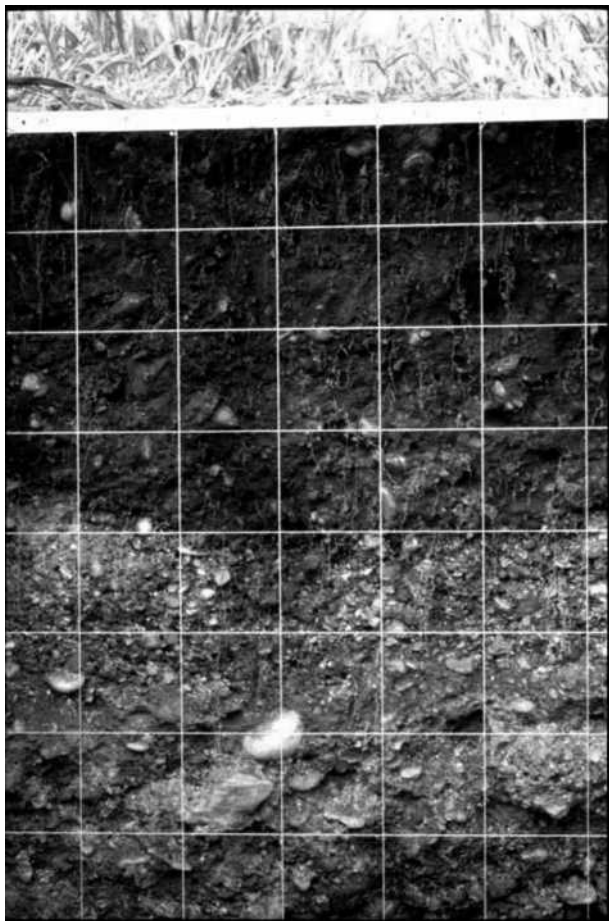


Figure 1. Visualization of flow patterns in soils using a dye tracer (Brilliant Blue FCF). Grid size is 10 cm.

SURFACE WATER, GROUNDWATER, AND VADOSE ZONE TRACERS

Fluorescent dyes are frequently used in surface and groundwater applications and, to some degree, in vadose

zone hydrology. Certain dyes, such as rhodamine WT and uranine, are used for surface water, groundwater, and vadose zone applications, whereas others, such as Brilliant Blue FCF, are exclusively used as vadose zone tracers. Many of the common dye tracers (Table 1) belong to the chemical class of the xanthene dyes. The structures of commonly used dyes are shown in Fig. 2.

Dye tracers have been used in measuring flow velocity, travel time, and dispersion in rivers and streams (19,20). Among the dyes commonly used as surface water tracers (Table 1), the most frequently used one is rhodamine WT (20–23). Uranine has been recognized as a good hydrological tracer, but its susceptibility to photochemical decay (17) is of concern in tracing surface water.

Dye tracers have also been used to study groundwater flow velocity, flow direction, hydraulic connections, and aquifer characteristics (4,5). Uranine and rhodamine WT are the two most commonly used tracers in groundwater studies (Table 1). However, these two dyes should not be used as cotracers because rhodamine WT degrades to carboxylic fluorescein, which may confound tracer quantification (13). Rhodamine WT is highly water soluble, easily visible and detectable, photochemically more stable than uranine, and has a moderate tendency for sorption (13). Commercially available tracer-grade rhodamine WT contains two isomers (Fig. 3), which have different sorption properties (24). The para-isomer of rhodamine WT sorbs less to different aquifer materials than did the meta-isomer (24,25). Consequently, the two isomers travel with different velocities in subsurface media, which lead to chromatographic separation (24). In groundwater tracer studies, like in surface water tracing, dye tracers can be easily detected or quantified in water samples using fluorometers or spectrophotometers. Methods and software for designing and analyzing tracer tests are available (13,26–28).

In the vadose zone, dyes are mainly used to delineate water flow patterns. Flow pathways in soils, sediments, and fractured rock have been visualized using dye tracers (29–33). Many dyes have been tested in search for an

Table 1. Dyes Commonly Used As Hydrological Tracers

Commercial Name	C.I. Nr.	C.I. Name	Chemical Class	Fluorescence	Maximum Excitation (nm) ^a	Maximum Emission (nm) ^a	Major Uses
Brilliant Blue FCF	42090	Food Blue 2	Triarylmethane	No	None	630 ^b	Vadose zone
Rhodamine WT	none	Acid Red 388	Xanthene	Yes	558 ^c	583 ^c	Surface water, groundwater, vadose zone
Sulforhodamine B	45100	Acid Red 52	Xanthene	Yes	560	584	Groundwater, vadose zone
Rhodamine B	45170	Basic Violet 10	Xanthene	Yes	555	582	Surface water, groundwater, vadose zone
Sulforhodamine G	45220	Acid Red 50	Xanthene	Yes	535	555	Groundwater
Uranine (Fluorescein)	45350	Acid Yellow 73	Xanthene	Yes	492	513	Groundwater, vadose zone
Eosine	45380	Acid Red 87	Xanthene	Yes	515	535	Groundwater, vadose zone
Methylene Blue	52015	Basic Blue 9	Thiazine	No	None	668 ^d	Vadose zone
Lissamine Yellow FF	56205	Acid Yellow 7	Aminoketone	Yes	422	512	Groundwater, vadose zone
Pyranine	59040	Solvent Green 7	Anthraquinone	Yes	460	512	Groundwater

^aSource: Field (27).

^bOur own data.

^cSutton et al. (24) reported the excitation maximum for both the para- and meta-isomers as 555 nm, and the emission maximum as 585 nm for the para-isomer and 588 nm for the meta-isomer.

^dSource: Merck (48).

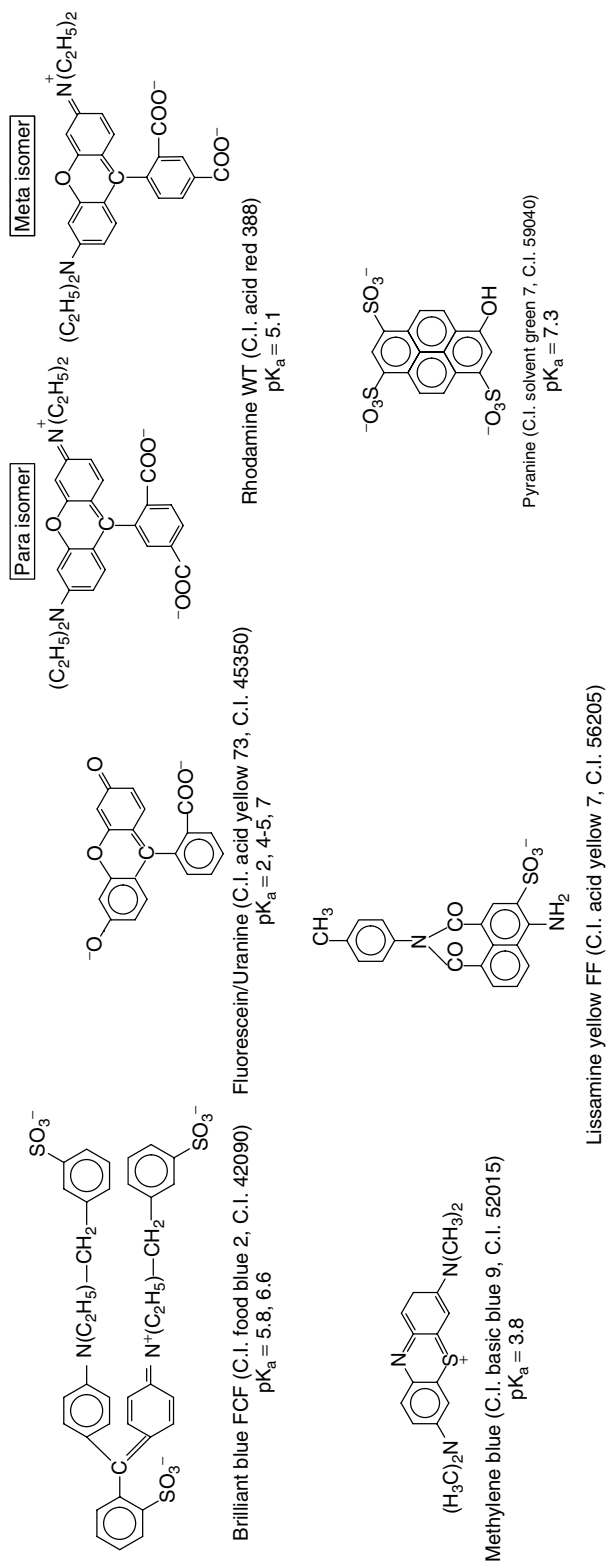


Figure 2. Structure of selected dye tracers. Dyes are shown in dissociated form. (Sources of the pK_a values are given in Ref. 6).

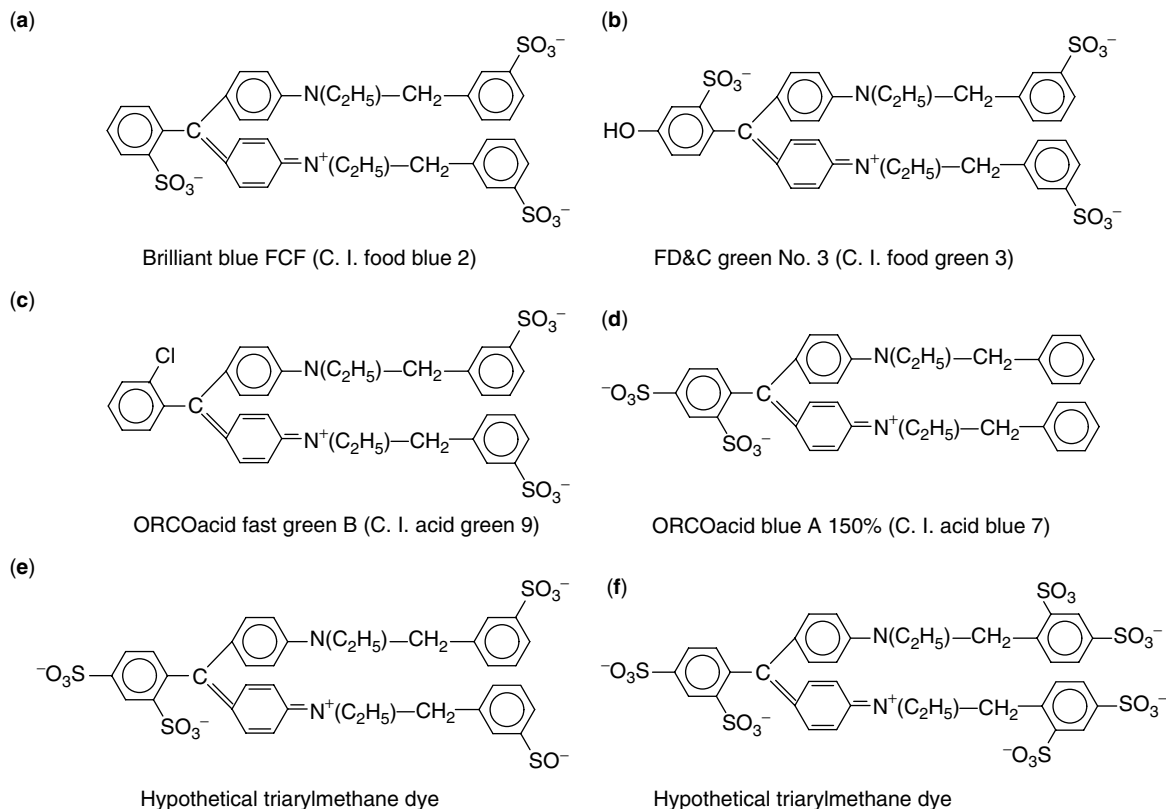


Figure 3. Test triarylmethane dyes that develop the QSAR model (a)–(d) and hypothetical structure of potential dye tracers (e)–(f). Dyes are shown in their anionic form.

optimal vadose zone dye tracer, and different dyes have been recommended (6). Most commonly used vadose zone tracers are listed in Table 1. Brilliant Blue FCF has gained acceptance as a good dye tracer for visualization of flow patterns (12,33) and solute transport in the vadose zone (34–36). In the vadose zone, dye tracer analysis is not as simple as in surface water or groundwater tracer studies, particularly if tracer concentrations are to be determined. Image analysis or fiber-optic spectroscopy can be used to measure tracer distributions in soil profiles (37,38).

LIMITATIONS IN USING DYES AS TRACERS

Most dyes are organic molecules, and their interactions with other materials in the subsurface are influenced by environmental conditions. Generally, dye tracers sorb to solid surfaces and the degree of sorption depends on surface properties and solution chemistry. Solubility, photochemical decay, absorption spectra, and fluorescence of dyes are often affected by environmental conditions, such as temperature, sunlight, acidity, and alkalinity. Thus, not only properties of the dyes but also of the environment in which dyes are to be applied often limit the use of dyes as tracers.

Sorption of dyes to subsurface media is one major limitation for using dyes to trace water flow pathways. Sorption causes dyes to move with a slower velocity than water. Some dyes can mimic the movement of certain

chemicals rather than the flow of water. For instance, rhodamine WT mimicked the movement of atrazine (39).

Dyes selected as hydrological tracers often contain functional groups, such as carboxylic and sulfonic acids, which contribute to high water solubility and decrease sorption (9,40). However, the functional groups cause dyes to have pH dependent properties. The properties of mineral surfaces may also change with pH, i.e., negatively charged surfaces may become neutral or positively charged as pH decreases, and sorption of anionic dyes may increase. Therefore, the sorption of dyes should be tested before dyes are applied as tracers.

Fluorescence of dyes may change under different environmental conditions. For instance, fluorescence intensity of rhodamine B increases with decreasing temperature (17). The presence of electron donating ions, such as chlorine, bromine, and iodine, in water samples as well as changes in solution pH can cause fluorescence quenching (10,19).

SELECTION OF DYE TRACERS FOR SPECIFIC USES

QSAR Approach as an Alternative to Experimental Screening

Screening is a basic step for selection of the most suitable dye tracers for specific uses, but experimental screening of thousands of commercially available dyes is not practical. An efficient technique (accurate, simple, fast, and inexpensive) is necessary to find the most suitable dye

tracer for a specific investigation. A promising screening technique is the use of QSAR.

QSARs relate the molecular structure of a chemical to its activity. Although this technique has been used extensively in pharmacology, it has also been applied to estimate environmental fate and risk of organic chemicals (41–44). QSAR models are based on calculated molecular descriptors and selected measured data that describe the property to be predicted. A statistical model then allows us to predict the properties of structurally similar chemicals that have not yet been experimentally tested.

QSAR Case Study Using Triarylmethane Dyes

We illustrate the use of QSAR for dye tracer screening using the example of the triarylmethane dyes. These dyes are often used as food dyes, and because they are highly water soluble, they have preferable characteristics as dye tracers (6). Brilliant Blue FCF, one member of this dye class, is commonly used as a vadose zone tracer. Other members, however, may be even better suited as dye tracers. We developed a QSAR model with triarylmethane dyes to predict their soil sorption characteristics. Four triarylmethane dyes were selected as a training set: Brilliant Blue FCF (C.I. Food Blue 2), FD&C Green No. 3 (C.I. Food Green 3), ORCOacid Blue A 150% (C.I. Acid Blue 7), and ORCOacid Fast Green B (C.I. Acid Green 9). These four dyes share the same molecular kernel but differ in numbers, types, and positions of functional groups [Fig. 3(a)–(d)].

We experimentally measured soil sorption parameters of the four dyes and used QSAR to relate these parameters to the structural properties of the dyes. Soil sorption was determined by batch sorption experiments similar to the ones described in German-Heins and Flury (16). A sandy soil (Vantage, WA), pH 8, and 0.01 M CaCl₂ solution were used for the sorption experiments. A Langmuir sorption isotherm was fitted to the experimental data to obtain the two adsorption parameters, the Langmuir coefficient K_L and the maximum adsorption A_m (Table 2), using a normal nonlinear least-squares method (45). The Langmuir isotherm describes the relation between sorbed (C_a) and aqueous concentrations (C_s) at equilibrium

as (45):

$$C_a = \frac{A_m K_L C_s}{1 + K_L C_s} \quad (1)$$

Structural properties (molecular descriptors) of the dyes were calculated using the MDL QSAR (version 2.1, 2002, MDL Information System, Inc., San Leandro, CA). The MDL QSAR program converts molecular structures to structural properties, such as molecular connectivity indices (MCIs), molecular volume, and surface area. Stepwise linear regression analyses were applied to select the descriptors that are well correlated to the experimental parameters (46,47). The statistical significance was assumed at $p \leq 0.05$.

The cross validation technique tested the predictability of the models. Randomization tests were performed to check the probability that correlation occurred by chance. The models that achieved the best quality of statistics were selected for estimation of each sorption parameter. The two QSAR models, one for estimation of K_L and another for estimation of A_m , were established as follows:

1. Langmuir coefficient (K_L) model:

$$K_L = -54.47(^9\chi_p) + 183.75 \quad (2)$$

where K_L has units of L/mmol and $^9\chi_p$ is the ninth-order simple path molecular connectivity index.

2. Maximum adsorption (A_m) model:

$$A_m = -45.72(^9\chi_p^v) + 35.88 \quad (3)$$

where A_m has units of mmol/kg and $^9\chi_p^v$ is the ninth-order valence path molecular connectivity index.

Prediction of Soil Sorption Using QSAR Models

Approximately 70 hypothetical molecules were created based on the structure of Brilliant Blue FCF, and their sorption parameters were estimated using the QSAR models (Equations 2 and 3). These molecules all shared the same molecular kernel as Brilliant Blue FCF but were different in number and position of SO₃ groups. The effects of different numbers and positions of SO₃ groups on soil sorption parameters, i.e., K_L and A_m values, of the new compounds were examined.

Table 2. Comparison of Langmuir Coefficient (K_L) and Maximum Adsorption (A_m) for Test and Hypothetical Triarylmethane Dyes

Triarylmethane Dyes	C.I. Nr.	Number of SO ₃ Groups	Langmuir Coefficient K_L (L/mmol)	Maximum Adsorption A_m (mmol/kg)
Test Triarylmethane Dyes			Experimental	
C. I. Food Blue 2	42053	3	5.29	0.42
C. I. Acid Blue 7	42080	2	10.1	2.99
C. I. Food Green 3	42090	3	3.94	0.30
C. I. Acid Green 9	42100	2	16.5	4.40
Hypothetical Triarylmethane Dyes			Predicted	
Dye set 1	none	1	20.9 to 37.8	5.8 to 11.2
Dye set 2	none	2	8.1 to 31.5	2.0 to 8.7
Dye set 3	none	3	−8.5 to 14.7	−2.9 to 4.1

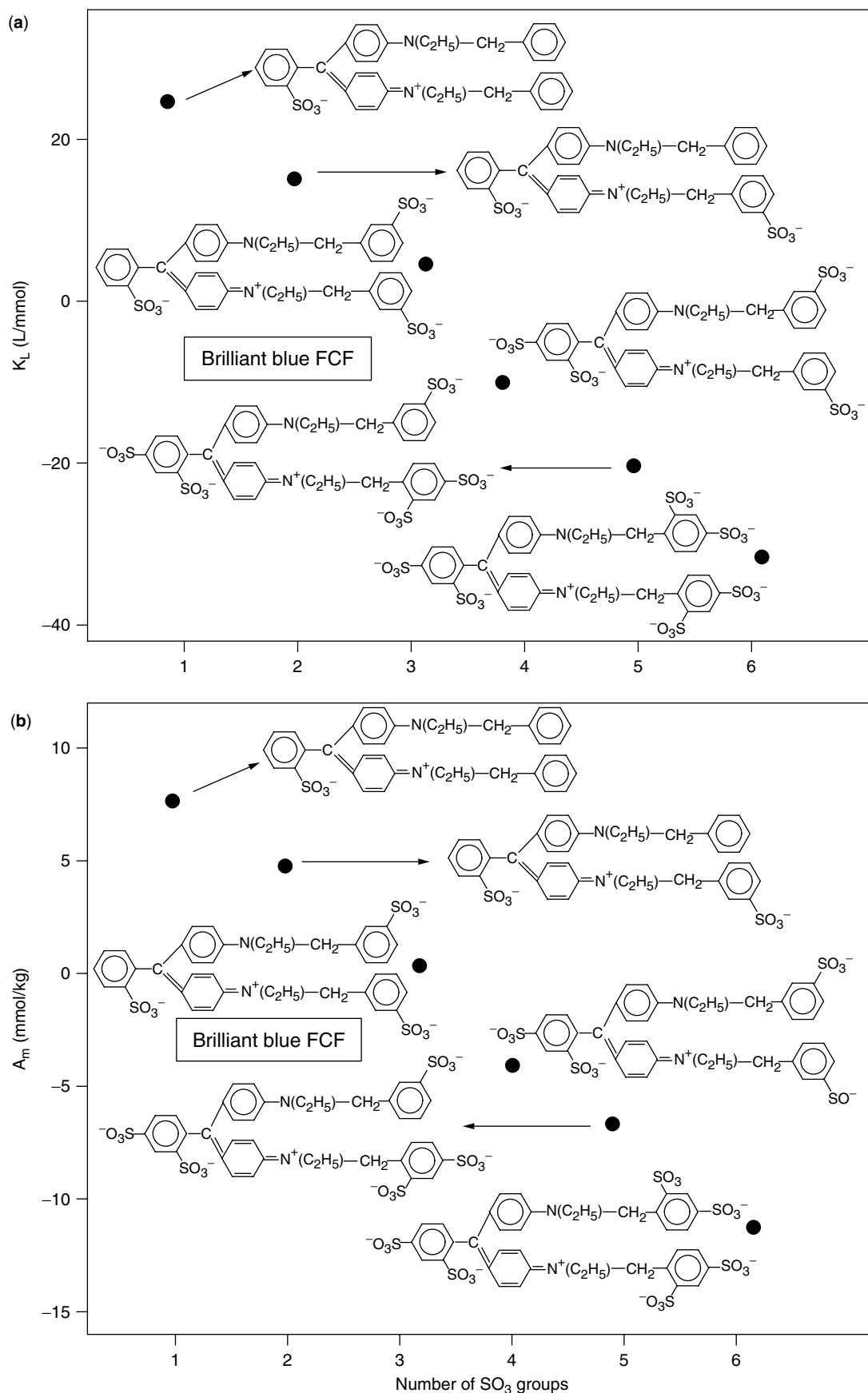


Figure 4. Changes in (a) Langmuir coefficient, K_L , and (b) adsorption maximum, A_m , as a function of the number of SO₃ groups on the molecular kernel of triarylmethane dyes.

The QSAR modeling indicates that the more SO₃ groups are attached to the molecular kernel, the smaller will be the soil sorption: Both K_L and A_m values decreased with the increasing number of SO₃ groups (Fig. 4). Negative K_L and A_m values were calculated by the models. Negative values are physically not possible, except in the case of ion exclusion. The predicted values should be considered as relative, rather than as absolute, measures for comparing the sorption of the chemicals.

QSAR modeling also examined the effects of the positions of the SO₃ groups at the molecular kernel. Three sets of hypothetical molecules were created, which contained one, two, or three SO₃ groups attached at different positions at the benzene rings of the triarylmethane kernel. Set 1, which contained one SO₃ group, consisted of six molecules, set 2 (two SO₃ groups) consisted of 22 molecules, and set 3 (three SO₃ groups) consisted of 31 molecules. The range of the predicted K_L and A_m values is listed in Table 2. The large variation in K_L and A_m values within each group of chemicals showed that the sorption parameters were strongly influenced by the positions of the functional groups.

Many dyes in sets 2 and 3 had lower K_L and A_m values than did the four test dyes. The hypothetical dyes with four to six SO₃ groups attached to triarylmethane kernel [Fig. 3(e) and (f)] had considerably smaller K_L and A_m values than did the test dyes. These hypothetical dyes are likely better conservative tracers than are any of the test dyes.

The K_L and A_m values of C.I. Food Green 3 were lower than were those of C.I. Food Blue 2 (Brilliant Blue FCF). Between these two readily available dyes, C.I. Food Green 3 may be a better tracer than Brilliant Blue FCF for hydrological investigations in the vadose zone.

SUMMARY

Dye tracers are frequently used in hydrological investigations. Although dyes have unique tracer characteristics, some limitations and problems are associated with using dyes as hydrological tracers. Most dyes sorb to subsurface media, so that tracer characteristics of dyes should be tested under the specific conditions under which dye tracing is to be conducted.

An accurate and cost-effective screening technique is necessary for selection of optimal dye tracers. QSARs offer a powerful tool for screening of a large number of dyes in a short time. We conducted a QSAR case study using triarylmethane dyes. The results of the QSAR modeling indicate that many hypothetical triarylmethane dyes have considerably lower sorption characteristics than do the triarylmethane dyes currently used as tracers, and they likely are good tracer candidates.

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FLOW-DURATION CURVES

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The flow-duration curve is a cumulative frequency curve that shows the percentage of time that specified stream's discharges were equaled or exceeded during a period of record (1). It combines in one curve all of the flow characteristics of a stream, ranging from flood flows to drought situations.

In sharp contrast to a hydrograph where time is on the x axis, chronology is not shown in flow-duration curves. If the period of record on which the curve is based is sufficiently long, it may be considered a probability curve and therefore, is used to estimate the percentage of time that a specified stream's discharge will be equaled or exceeded in the future.

INTRODUCTION

The temporal sequence of flows is not included in the preparation of flow-duration curves, so one cannot tell from the curve itself whether periods of low or high flow occurred during one dry or wet period or were scattered over several years (2). However, the curve is very useful

for studying the flow characteristics of a stream for the entire range of discharge that can vary across several orders of magnitude.

Although flow-duration curves can be based on weekly or monthly averages, the greatest accuracy is reached in using the day as the unit of time. Therefore, it is recommended that flow-duration curves used in hydrologic analysis be based on mean daily discharges for the period of record.

As streamflow generally has a log-normal distribution, flow-duration curves are usually plotted on logarithmic probability graphs where the x axis represents the percentage probability that a given discharge was equaled or exceeded during a given period and the y axis indicates discharge in either cubic feet/second or cubic meters/second. A sample flow-duration curve for Bowie Creek near Hattiesburg, Mississippi is shown in Fig. 1. Note that the x axis ranges from 0.05 to 99.99% and the y axis is scaled logarithmically from 100 to 10,000 cubic feet/second (two orders of magnitude).

THE RELATIONSHIP BETWEEN CURVE SHAPE AND GEOHYDROLOGY

It is widely recognized that the shape of the flow-duration curve provides very useful information about

the geohydrologic characteristics of watersheds. Steeply sloping curves indicate “flashy” streams where the flow is largely from direct runoff and where there is limited groundwater storage. For example, the flow-duration curves for the South Branch Raritan and Great Egg Harbor Rivers in New Jersey for the 1931–1960 time period are shown in Fig. 2. The South Branch Raritan River in North Jersey drains consolidated rock formations such as Precambrian granites and Paleozoic limestones. In contrast, Great Egg Harbor River in the unconsolidated formations of the Coastal Plain of South Jersey is underlain by thick seaward-dipping deposits of sand (3). The geohydrologic differences between the consolidated rocks of North Jersey and the unconsolidated sedimentary formations of the Coastal Plain are illustrated by the different slopes in Fig. 2. First, discharge on the y axis for both rivers has been equilibrated by using cubic feet per second/square mile values to make the flow independent of watershed size (4). Second, note in Fig. 2, that the discharge values are higher for the South Branch Raritan River at high flow conditions (0.01–20% exceedance values) and lower at low flow conditions (80–99.5% exceedance values). This difference is to be expected as the groundwater storage is much greater in the Coastal Plain than in the consolidated rock formations of northern New Jersey.

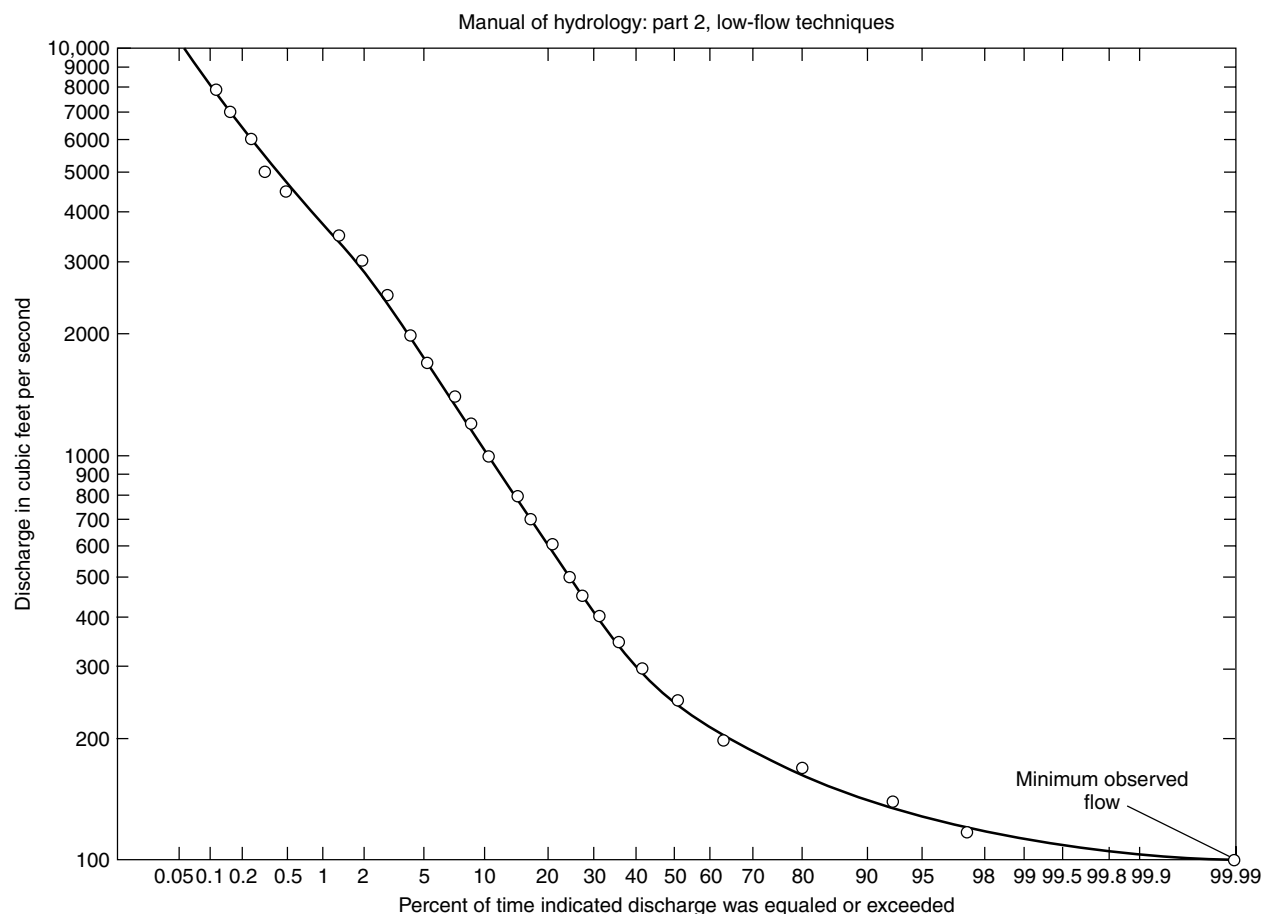


Figure 1. Duration curve of daily flow, Bowle Creek near Hattiesburg, Mississippi, 1939–1948.

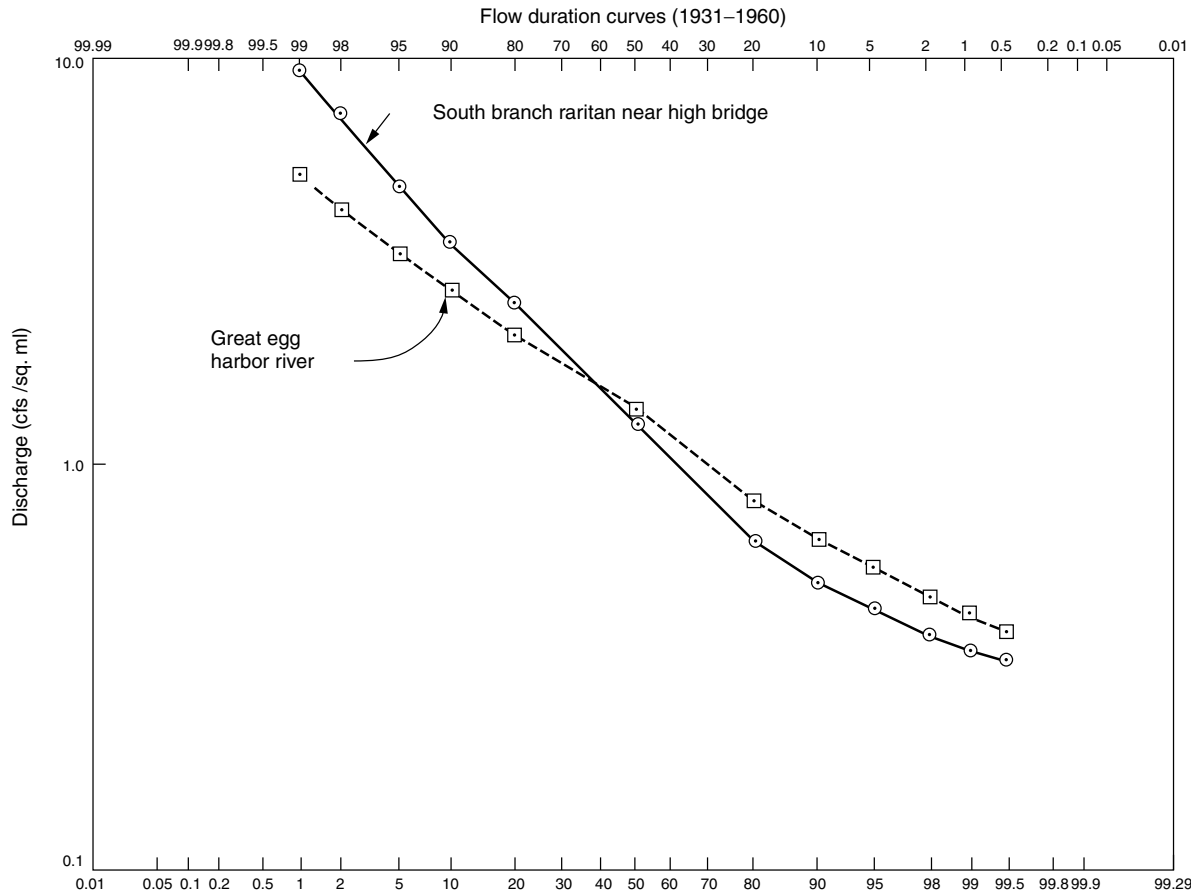


Figure 2. Percentage of time indicated discharge was equaled or exceeded.

An even better example of differential geohydrology and its effect on flow-duration curves is shown in Fig. 3. Both the South Branch Raritan River and Stony Brook at Princeton in New Jersey are underlain by consolidated rock formations. However, the geologic formations within the Stony Brook watershed include a large proportion of Lockatong argillite, a very tight, poorly fractured rock that is notorious for its poor groundwater yields and limited water storage. As in Fig. 2, discharge on the y axis has been equilibrated, in this case by using (mgd) million gallon per day/square mile values.

Note the precipitous decline of the slope of the curve for Stony Brook in Fig. 3, indicating the strong effects of the minimal amounts of groundwater in storage in the argillite formation. The flow-duration curve data were plotted on log-probability graphs, so the decline in expected flow is even more pronounced.

VARIABILITY INDEXES

One of the major characteristics of streamflow is its variability. This variability reflects the variability in precipitation as modified by the physical characteristics of the watershed. These physical characteristics include the differential geohydrology of consolidated and unconsolidated rock formations and the water storage available in lakes, swamps, and wetlands. Another factor that will become of increasing importance in streamflow variability

is the anthropogenic factor as exemplified by the effects of urbanization on the hydrologic cycle. Streams are subject to more regulation in the form of water supply diversions, interbasin transfers of both raw water and treated effluent, and low flow release requirements, so the natural flow of a stream may become very difficult to measure, particularly in heavily urbanized smaller basins.

One quantitative measure of streamflow variability is the slope of the flow-duration curve itself. The steeper the slope, the greater the variability. A numerical index of variability was introduced by Lane and Lei (5) that is defined as the standard deviation of the logarithms of stream discharge at 10 points on the curve between the 5% and 95% exceedance values (5, 15, 25, ... 95%).

Miller (6) noted that many flow-duration curves tend to be nearly straight lines on log-probability paper between the 20 and 80% exceedance points. The curves above and below these points depart strongly from a straight line and would not be suitable for a numerical index. Thus, Miller (6) suggested using a simple index by dividing the discharge at the 20% point on the curve by the discharge at the 80% point. Flashy streams have high indexes, whereas streams of relatively uniform flows (such as those in the Coastal Plain) have low indexes. One additional benefit of this easily calculated index is that the resulting ratio is dimensionless, thereby facilitating the comparison of streamflow variability in varied locations.

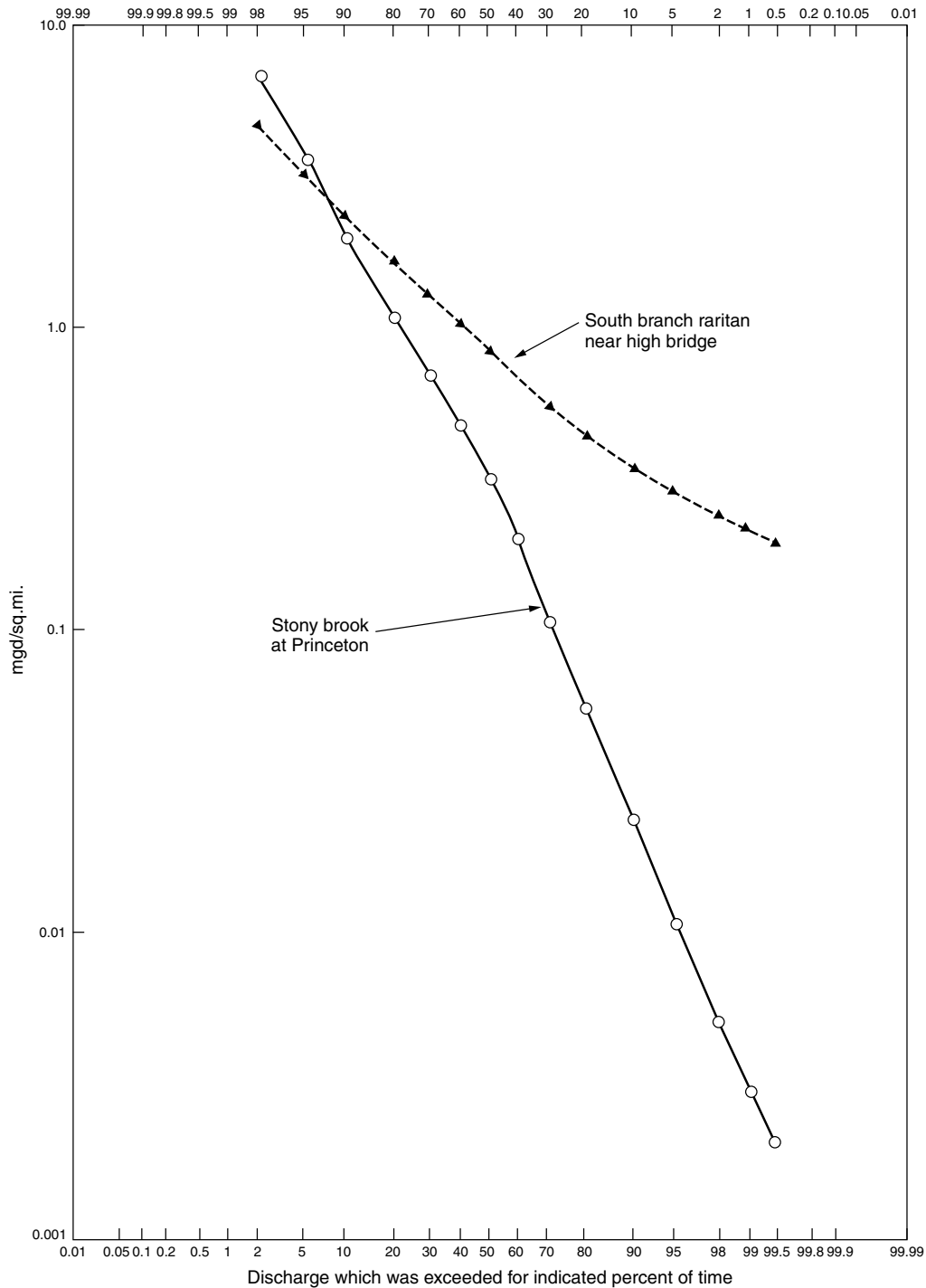


Figure 3. Flow-duration curves; Stony Brook at Princeton, 1954–1975; South Branch Raritan near High Bridge, 1919–1975.

SUMMARY

A flow-duration curve is a graph, usually plotted on log-probability paper, that shows the percentage of time that streamflow at a gaging station is either equaled or exceeded. Important information about the hydrologic and geologic characteristics of a watershed is revealed in the shape of the curve. Curves that slope steeply represent streams of very variable flows that reflect large inputs of overland runoff. Conversely, curves of gentler slopes are

indicative of basins that have large amounts of groundwater in storage that result in flows of higher constancy.

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ENVIRONMENTAL FLOWS

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The ecological integrity of riverine ecosystems depends on their natural dynamic character (1). Sustaining the natural functioning of aquatic and water-dependent ecosystems in the face of growing human demands is a major challenge (2). Its solution requires the allocation of water to protect aquatic ecosystems from the impacts of abstraction and river flow regulation (see Regulated Rivers), and this is one of five facets to freshwater management for the twenty-first century (see Water in History). It also requires acceptance of a multiuse ethic to provide flexible water budgets to support ecological functions in both dry and wet years (3).

Rivers are the arteries of fluvial hydrosystems comprising the river itself, riparian zone, floodplain, and alluvial aquifer. These four-dimensional systems are influenced by longitudinal processes, vertical and lateral fluxes, and strong temporal changes. Flow is the driver. Flow defines the environmental domains within which biological communities develop, including the vertical exchanges between surface and groundwater. High flows interact with the sediment load, and also with downed trees and driftwood (4), to shape the channel and floodplain morphology and to structure the complex mosaic of physical habitats that comprise a river corridor.

A major constraint to the advancement of tools for allocating flow to meet in-river needs was the lack of scientifically based models (5). In the 1960s and 1970s, early attempts to set *instream flows* for rivers focused on the annual minimum flow expressed as a hydrological statistic, commonly as either a flow duration statistic (such as the 95th percentile flow) or as a fixed percentage of the average daily flow (ADF), with several studies proposing 20% ADF to protect aquatic habitat in streams.

From the early 1980s, more complex approaches to determine instream flows were introduced that make the assumption that physical habitat attributes provide an index of suitability for biota. The most well known of these flow-habitat methodologies, Physical Habitat Simulation (PHABSIM), integrates the changing hydraulic conditions with discharge and the habitat preferences of one or more selected species. The method relies on three principles:

The chosen species exhibits preferences within a range of habitat conditions that it can tolerate; these ranges can be defined for each species; and the area of stream providing these conditions can be quantified as a function of discharge and channel structure. In the majority of cases, these methodologies developed instream-flow guidelines that focused on the needs of a single species, usually a salmon or trout, although more advanced approaches considered the needs of different life stages. More recent advances of these approaches include, for example, assessment of flows (1) to encourage the upstream migration of salmon from the estuary to the spawning grounds, (2) to provide good quality habitat for spawning, and (3) to maintain the intragravel environment for egg development over winter for fry at emergence in late winter and for juvenile fish in spring and summer.

By the early 1990s, the science and management of *regulated rivers* had expanded from the determination of *instream flows* to *environmental flows*. Many schemes now applied more complex flow-habitat models to address wider issues than the instream needs, for example, the *hydraulic habitats* of a single species. These new approaches address the sustainability of communities and ecosystems. They address the access of aquatic biota to seasonal floodplain and riparian habitats as well as the need for high flows to sustain the geomorphological dynamics of the river corridor and floodplain habitats (6). They provide more sophisticated approaches to setting minimum flows and enable advancement of an ecologically acceptable flow regime concept (7,8), which recognizes that a set of minimum flow constraints does not provide sufficient protection for river ecosystems. Different life stages and different species benefit from different flows at different times of the year, and in different years. Rivers must be protected in wet years as well as drought years because high flows provide optimum conditions for some species and are also responsible for sustaining the quality and diversity of in-channel and riparian habitats.

The basic environmental principles needed to formulate policy decisions and management approaches on environmental flows have been summarized by Naiman et al. (9). These focus on the need to sustain *flow variability* that mimics the natural, climatically driven variability of flows at least from season to season and from year to year, if not from day to day. The two fundamental general principles are:

1. The natural flow regime shapes the evolution of aquatic biota and ecological processes.
2. Every river has a characteristic flow regime and an associated biotic community.

From these were developed four specific principles for advancing the provision of environmental flows (10):

1. Flow is a major determinant of physical habitat in rivers, which in turn is a major determinant of biotic composition.
2. Maintenance of the natural patterns of connectivity between habitats along a river and between a river and its riparian zone and floodplain is essential to the viability of populations of many riverine species.

3. Aquatic species have evolved life history strategies primarily in response to the natural flow regime and the habitats that are available at different times of the year and in both wet and dry years.
4. The invasion and success of exotic and introduced species along river corridors are facilitated by regulation of the flow regime, especially with the loss of natural wet-dry cycles.

In addressing the issues of environmental flows, water managers must become water-and-habitat managers, and holistic management strategies are being developed to support their activities (6). However, a continuing failure by policymakers to give due recognition to the array of goods, services, and other benefits provided by aquatic ecosystems (9), to the complexity of ownership and rights of access to these, and of how to integrate livelihood issues into water and ecosystem resource-management decisions remains a major issue.

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EUTROPHICATION (EXCESSIVE FERTILIZATION)

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INTRODUCTION

“Eutrophication” is the process of a waterbody becoming increasingly rich in aquatic plant life such as algae and aquatic macrophytes (water weeds). It is driven by the increasing input of aquatic plant nutrients, principally

nitrogen and phosphorus, from natural and anthropogenic sources. Although natural eutrophication takes place over geologic time, activities of people that increase the aquatic plant nutrient inputs to waterbodies can rapidly accelerate this process and cause cultural eutrophication. Thus, the term “eutrophication” has become synonymous with “excessive fertilization” or the input of sufficient amounts of aquatic plant nutrients to cause the growth of excessive amounts of algae and/or aquatic macrophytes in a waterbody such that beneficial uses of the waterbody (i.e., water quality) are impaired. Beneficial uses of waterbodies that stand to be impaired by the presence of excessive amounts of aquatic plant life include domestic and industrial water supply, recreation, and fisheries.

Because of the public health and environmental quality significance of these water quality impairments, myriad strategies have been advanced to evaluate and regulate excessive fertilization and nutrient input to waterbodies, with varying degrees of technical validity and demonstrated effectiveness. This chapter reviews what should be considered in assessing the impacts of nutrients that are added to a waterbody on the waterbody’s eutrophication-related water quality. References are provided to more detailed discussions of the issues covered.

IMPACTS OF EXCESSIVE FERTILIZATION ON WATER QUALITY

The excessive fertilization of waterbodies is a long-standing, well-recognized water quality problem throughout the United States and other countries. It is manifested as excessive growths of planktonic (suspended) algae, attached algae, and aquatic macrophytes (water weeds). Aquatic macrophytes can be floating forms such as water hyacinth or duckweed or attached-emergent forms. Water quality problems caused by these growths, discussed in detail by Lee (1), are summarized below.

Domestic Water Supplies

When raw water supplies contain large amounts of algae and some other aquatic plants, the cost of treatment increases and the quality of the product may be diminished. Planktonic algae can shorten filter runs. They can also release organic compounds that cause tastes and odors and, in some instances, serve as trihalomethane (THM) and haloacetic acid (HAA) precursors. THMs are chloroform and chloroform-like compounds; HAAs are low-molecular-weight chlorinated organic acids. These compounds are produced when the precursors react with chlorine during the disinfection process and are regulated as human carcinogens.

Violations of Water Quality Standards

Excessively fertile waterbodies can exhibit marked diel (over a 24-hr day) changes in pH and dissolved oxygen concentrations that can result in repeated short-term violations of water quality standards. During daylight, algal photosynthesis removes CO₂ from the water, which increases the pH; algal respiration in the night releases CO₂ and lowers the pH. In late afternoons, the pH

of excessively fertile water can be found to exceed the water quality standard for pH. Similarly, algae produce oxygen during photosynthesis, but they consume it during respiration. Just before sunrise, after sufficient nighttime algal, bacterial, and other organism respiration, dissolved oxygen concentrations can be below water quality standards for protection of fish and other aquatic life. Excessively fertile waterbodies that thermally stratify (develop a thermocline) often exhibit dissolved oxygen depletion below the thermocline because of bacterial respiration of dead algae. Richards (2) showed that one phosphorus atom, when converted to an algal cell that subsequently dies, can consume 276 oxygen atoms as part of the decay process.

Toxic Algae

One major stimuli for the U.S. EPA's recently increased attention to excessive fertilization is the *Pfiesteria* problem in Chesapeake Bay (3); fish kills occurred there because of the presence of toxic algae. Fish kills associated with toxic algae have occurred in various waterbodies around the world, including off the west coast of Florida, for many years. In addition, blue-green algae at times excrete toxins that are known to kill livestock and other animals that consume the water.

Impaired Recreation and Aesthetics

Excessive growths of attached algae and aquatic macrophytes can impair swimming, boating, and fishing by interfering with water contact. Severe odor problems can also be caused by decaying algae, water weeds, and algal scums.

Water clarity—defined by the depth of the waterbody at which the bottom sediments can be seen from the surface—is an aesthetic quality that is compromised by

eutrophication. Waterbodies with high degrees of clarity (i.e., the bottom can be seen at depths of 20 or more feet) have low planktonic algal content; in more eutrophic waterbodies, the sediments can only be seen at a depth of a few feet. The greenness of water, which contributes to diminished water clarity and is caused by the presence of algae, can be quantified by measurement of planktonic algal chlorophyll. Inorganic turbidity also diminishes water clarity and can influence the perception of greenness of a waterbody. Often, high levels of planktonic algal chlorophyll can be present in a shallow waterbody or river without the public's perceiving it to be excessively fertile, if the water is brown because of inorganic turbidity.

Impact on Fisheries

As illustrated in Fig. 1, fertilization increases total fish production (biomass). However, as Lee and Jones (4) discussed, it can adversely affect the production of desirable types of fish, especially at high fertilization levels. In stratified waterbodies, algae grow in surface waters, die, and settle to the hypolimnion (bottom layer) where they are decomposed. As noted above, the oxygen demand created by algal decomposition can be sufficient in eutrophic waterbodies to deplete the hypolimnetic oxygen, which means that the desirable coldwater fish (e.g., salmonids, trout) that normally inhabit the cooler hypolimnion cannot survive there because of insufficient oxygen. Thus, the higher fish production characteristic of highly eutrophic waterbodies is typically dominated by rough fish, such as carp, which can tolerate lower dissolved oxygen levels.

Shallow Water Habitat

Emergent aquatic vegetation in shallow waters provides important habitat for various forms of aquatic life. As

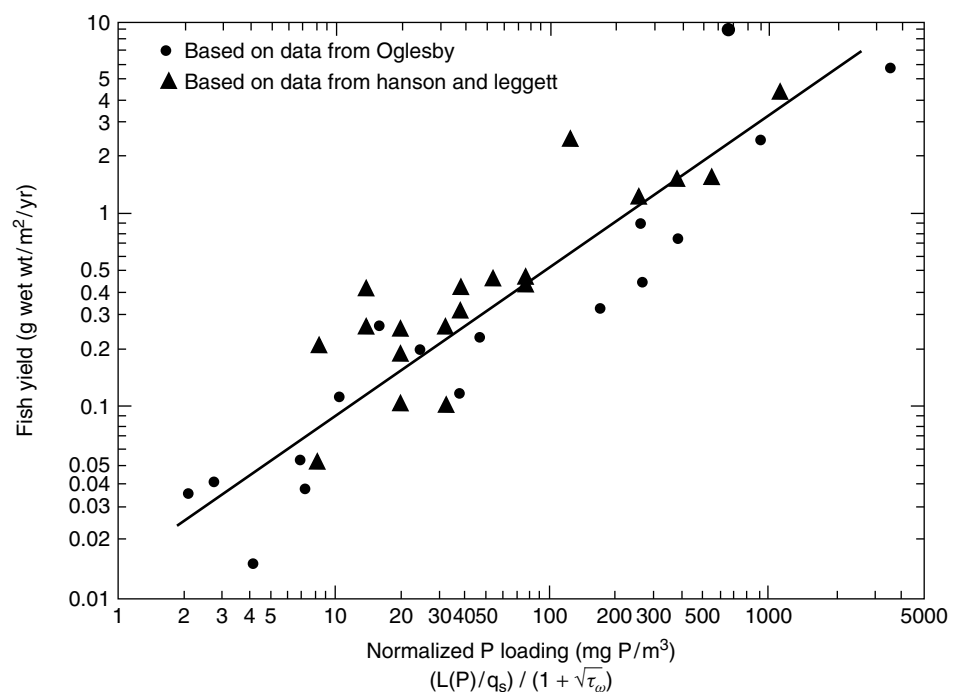


Figure 1. Relationship between normalized P load and fish yield [from Lee and Jones (4)].

discussed by Lee (1), increased planktonic algal growth reduces light penetration (water clarity), which in turn inhibits the growth of emergent vegetation. This process can result in loss of significant aquatic life habitat.

Overall Impacts

Excessive fertilization is one of the most important causes of water quality impairment of waterbodies. In its last National Water Quality Inventory, the U.S. EPA (3) listed nutrients as the leading cause of impairment of lakes and reservoirs.

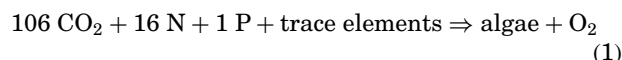
CONTROLLING EXCESSIVE FERTILIZATION

Algae and other aquatic plants require a wide variety of chemical constituents, light, and appropriate temperatures to grow. Of those factors, however, only nutrient input is amenable to sufficient control to effect a meaningful decrease in algal and aquatic plant biomass to reduce the adverse impacts of excessive fertilization. The issues of which nutrient(s) should be controlled, sources of the nutrient, what type of and how much control is needed, and the positive impacts of the control must be addressed in a eutrophication management program.

Limiting Nutrient

For managing algal populations, the primary focus should be on control of the nutrient that is present in the least amount compared with algal needs, i.e., the limiting nutrient. Increasing or reducing the amount of that nutrient available to algae will affect an increase or a decrease in the algal biomass that can be sustained. This process is illustrated in Fig. 2, which shows that additional growth occurs in response to additional input of the limiting nutrient up to the point at which it is present in greater amounts than can be used. Nitrogen and phosphorus are the nutrients that typically limit algal growth. Phosphorus is more often the limiting nutrient in freshwater waterbodies, whereas nitrogen is often the limiting nutrient in marine waters. Although the potassium content of some soils can limit the growth of terrestrial plants, potassium is not an element that limits aquatic plant growth.

To determine which nutrient is limiting algal growth in a particular waterbody, some have relied on the comparison of the concentrations of nitrogen and phosphorus to the "Redfield" stoichiometric ratio of these elements in algae (16:1 atomic basis or 7.5:1 mass basis) shown in Eq. 1.



It is presumed that if the ratio is smaller than this, N would be limiting, and vice-versa, which can give misleading results and lead to unreliable nutrient control measures because whatever the "ratio," either or both could be present in ample amounts for algal growth (5). Rather, it is the concentration of algal-available forms of nutrients at peak biomass—when the algal growth is being limited—that should be assessed. If the concentration

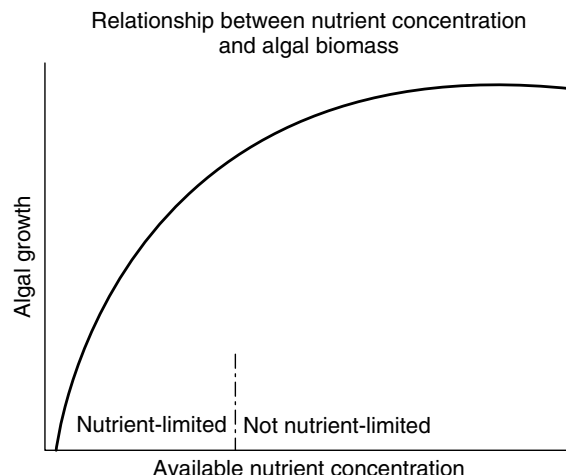


Figure 2. Relationship between nutrient concentration and algal biomass [from Lee and Jones-Lee (4)].

of either has been decreased by its utilization to below growth-rate-limiting concentration, reasonable certainty exists that that nutrient is limiting algal growth.

Typically, growth-rate-limiting concentrations for phosphorus are on the order of 2 to 8 $\mu\text{g/L}$ available P, and for nitrogen, 15 to 20 $\mu\text{g/L}$ available N. It is important to recognize, however, that even growth rate-limiting concentrations can support appreciable algal biomass if sufficient time is available for algal growth to occur. Furthermore, in many highly fertile waterbodies, neither nitrogen nor phosphorus is limiting algal growth. Both can be present above growth-rate-limiting concentrations—i.e., on the plateau of the algal growth-nutrient concentration relationship in Fig. 2.

AVAILABILITY OF NUTRIENTS

Nitrogen and phosphorus exist in aquatic systems in many different forms, only some of which can be used by algae and aquatic plants. Therefore, in assessing the limiting nutrient in a waterbody or evaluating the control of nutrient input to a waterbody, it is essential to consider the forms in which the N and P exist in the loading sources and waterbody. Algal available forms of nitrogen are nitrate, nitrite, ammonia, and after conversion to ammonia, some of the organic nitrogen. The fraction of the organic nitrogen that is available is site-specific and depends on its source and age. Under limited circumstances, some blue-green algae can fix (utilize) atmospheric nitrogen gas (N_2) that is dissolved in water and use it as a source of nitrogen for growth. Soluble orthophosphate is the form of phosphorus that is available to support algal growth. Most particulate phosphorus and organophosphorus compounds, and oxygen-phosphorus polymer chain and ring compounds (condensed phosphates), do not support algal growth.

In developing nutrient criteria, the U.S. EPA (6,7) has been focusing on total phosphorus rather than on algal-available forms. This approach can misdirect control programs to sources whose control will not result in cost-effective improvements in eutrophication-related water

quality. For example, it was well established many years ago that most of the particulate phosphorus in agricultural and urban stormwater runoff is not available to support algal growth. Lee et al. (8) reported on their extensive research as well as on the findings of others on this topic in a review of these issues for the International Joint Commission for the Great Lakes. From both short-term and long-term (one-year) tests, they found that the algal available P in agricultural and urban runoff can be estimated as the sum of soluble ortho-P and about 20% of the particulate P. Thus, most particulate P in agriculture and urban stormwater runoff from a variety of sources is not available for algal growth.

The lack of availability of much phosphorus in soils is well known to the agricultural community, which finds that total P in soils is not a reliable measure of plant-available P. As discussed by Lee and Jones-Lee (9), nutrient criteria for regulating agricultural and urban stormwater runoff should be based on soluble orthophosphate and nitrate plus ammonia plus about 20% of the particulate P and N. However, if the source of the P and N is algae, then most of the total N and total P will be mineralized and in time will become available to support algal growth.

SOURCES AND CONTROL OF ALGAL NUTRIENTS

Domestic Wastewater Discharges

Lee and Jones (10) reviewed the North American experience in controlling the excessive fertilization of waterbodies. They reported that domestic wastewater discharges are one of the most significant and controllable sources of available nutrients contributing to eutrophication. To control phosphorus from this source, tertiary treatment of the wastewaters is commonly practiced. Chemical treatment using alum (aluminum sulfate) typically costs a few cents per person per day for the population served by the treatment plant. Enhanced biological treatment of domestic wastewaters may also significantly reduce the phosphorus content of domestic wastewaters. Typically, either chemical or enhanced biological treatment can reduce the phosphorus concentration in domestic wastewater effluent by 90% to 95%. The authors estimate that the domestic wastewaters of more than 100 million people in the world are treated for phosphorus removal to reduce the excessive fertilization of the waterbodies receiving the wastewater discharges.

Nitrogen can also be removed from domestic wastewaters although not as readily as phosphorus. Nitrogen removal generally involves nitrification of the ammonia and organic nitrogen to nitrate, followed by denitrification. The cost is typically five to ten times greater than for phosphorus removal. Although phosphorus control in domestic wastewaters is widely practiced, nitrogen control has only been implemented to a limited extent because of the higher cost and because for most freshwater waterbodies, phosphorus control is the more effective way to control excessive fertilization.

Land Runoff

Another source of nutrients for waterbodies is runoff from land. Based on the U.S. Organization for Economic Cooperation and Development (OECD) Eutrophication Study data for about 100 waterbodies' watersheds located across the United States, Rast and Lee (11) determined nutrient export coefficients for the main categories of land use. Shown in Table 1, these coefficients define the mass of N and P that runs off a unit area of watershed land annually.

Although the export coefficients for a given watershed depend on the particular setting, the values in Table 1 have shown reliability in several areas for estimating the potential significance of various types of land use in contributing nitrogen and phosphorus from a watershed. More specific nutrient export coefficients for agricultural lands should be evaluated based on soil characteristics, types of crops grown, and other factors that tend to influence the amount of nitrogen and phosphorus exported from the land. Although these coefficients are for total N and total P, when used in the Vollenweider–OECD eutrophication modeling approach discussed subsequently, the availability of the loading is taken into account.

Nutrient Runoff Control BMPs

Controlling nitrogen and phosphorus in runoff from rural land has not been highly successful. Sharpley (12) reviewed the experience in trying to achieve a 40% reduction in nitrogen and phosphorus loads from agricultural lands in the Chesapeake Bay watershed. He indicated that limited progress has been made toward achieving that goal after about 15 years of effort. Similarly, Logan (13) reported that little progress has been made in effectively controlling phosphorus from agricultural runoff in the Lake Erie watershed.

Sprague et al. (14) reviewed factors that affect nutrient trends in major rivers of the Chesapeake Bay watershed. They noted the difficulty discerning major changes in the contribution of nutrients from agricultural lands in the watershed caused by year-to-year variability in nutrient export. This variability is related to several factors, including climate. They indicated that one of the principal methods for nutrient reduction from agricultural lands has been land retirement, i.e., termination of agricultural activities on the land.

Various "best management practices" (BMPs) have been implemented to control nutrient export from

Table 1. Watershed Nutrient Export Coefficients [from Rast and Lee (11)]

Land Use	Export Coefficients (g/m ² /y)		
	Total Phosphorus	Total Nitrogen	
Urban	0.10	0.5	0.25*
Rural/Agriculture	0.05	0.5	0.20*
Forest	0.01	0.3	0.10*
Other:			
Rainfall	0.02	0.8	
Dry Fallout	0.08	1.6	

*Describe nitrogen loadings for waterbodies in Western United States.

agricultural activities, including grassy strips, buffer lands, altering fertilizer applications, and so on. The U.S. EPA (15) discussed the current information on BMPs to control potential pollutants derived from agricultural lands. Although claims are made as to their effectiveness, it is evident from the U.S. EPA review and the authors' experience that there is a lack of quantitative understanding of the cost-effectiveness of BMPs for control of nutrients from agricultural activities (16). Quantitative studies are urgently needed to determine how various BMPs influence phosphorus and nitrogen export from the land, efficacy for controlling eutrophication, as well as costs associated with controlling phosphorus export to various degrees (e.g., 25%, 50%, and 75%). This information then needs to be viewed in the context of what agricultural interests of various types can afford relative to market prices, including issues of foreign competition. Maintaining agriculture through subsidies is a long-standing tradition in the United States. The control of nutrients from agricultural lands for the benefit of downstream waterbody users may also become one of the subsidy issues that will need to be considered to keep agriculture viable (although subsidized) in many parts of the United States.

Importance of Light Penetration

Algal growth in almost all waterbodies is light-limited to some extent. Turbidity and natural color diminish the penetrability of light into a waterbody, which affects the extent to which algae can use available nutrients. In fertile waterbodies, where the presence of abundant planktonic algae reduces the penetration of light further by self-shading, algae can photosynthesize only in the upper few feet of water. It is important to understand the influence of inorganic turbidity and natural color on the coupling between nutrient loads and eutrophication-related water quality. Although erosion from a waterbody's watershed may increase the nutrient load, it also increases the turbidity in the waterbody, which in turn decreases light penetration and thereby slows algal growth. Thus, control of erosion in a waterbody's watershed can result in greater algal growth for the same nutrient concentration than would occur if the waters were still turbid from erosion in the watershed.

Issues That Need to be Considered in Developing Appropriate Nutrient Control Programs

Several key issues need to be considered and evaluated in formulating nutrient control programs, the most important of which is the relationship between nutrient load and eutrophication-related water quality in the waterbody of concern. Each waterbody has its water quality-related load—a response relationship that needs to be defined.

First, the nature of the water quality impairment needs to be defined, which includes defining what the problem is (e.g., recreation impairment, aesthetics, tastes, and odors), when the water quality problems occur (e.g., summer, fall, winter, and spring), how eutrophication is manifested (planktonic algae, attached algae, and macrophytes),

and the desired eutrophication-related water quality characteristics. Next, the limiting nutrient during the period of concern and the primary sources of that nutrient should be determined. Each source should be evaluated for the availability of nutrients, the controllability of the available nutrients, and the cost of implementing and maintaining the control strategy. Finally, a reliable modeling approach needs to be applied to estimate the improvement in eutrophication-related water quality that would be effected by the estimated expenditures for the potentially viable control options.

Desired Nutrient-Related Water Quality

The first step in developing appropriate nutrient load criteria is to identify the eutrophication-related water quality problem as well as the desired outcome of management for the waterbody. Types of problem/solution goals that may be identified include, as discussed above, preventing violations of average or worst-case diel DO or pH standards, controlling algae-caused domestic water supply raw water quality problems (e.g., controlling tastes and odors, lengthening filter runs, reducing THMs, etc.), or increasing water clarity (Secchi depth). This evaluation should be done through a public process conducted by the regulatory agency because the public's perception of eutrophication-related water quality can be site-specific. In those areas where there are numerous waterbodies with marked differences in lake water clarity, for example, the public has the opportunity to compare waterbodies that are green with those that are clearer. There, the public's perception of high water quality is different from that in areas where all waters have the same general greenness because of planktonic algae.

Nutrient control must be undertaken with appropriate consideration of factors that govern how the nutrient loading is used within the specific waterbody. Eutrophication modeling can integrate these factors to relate nutrient load to eutrophication-related water quality response. Basically two types of eutrophication models exist:

- An empirical, statistical model, such as the Vollenweider–OECD eutrophication model discussed subsequently herein, developed from a large database quantifying how nutrient concentrations or loads relate to the nutrient-related water quality characteristics of the waterbody.
- Deterministic models, in which differential equations can describe the primary rate processes that relate nutrient concentrations/loads to algal biomass.

Deterministic models have several drawbacks for use in eutrophication management. Because of the number of equations incorporated into a deterministic model, no unique solution exists to the model. "Tuning" the model to match the nutrient loads and eutrophication condition in the waterbody of interest at the outset may not properly represent the conditions and response after nutrient load alteration. Thus, its ability to reliably meet the goal of management evaluation, i.e., predicting the benefit to be gained by management options, is limited.

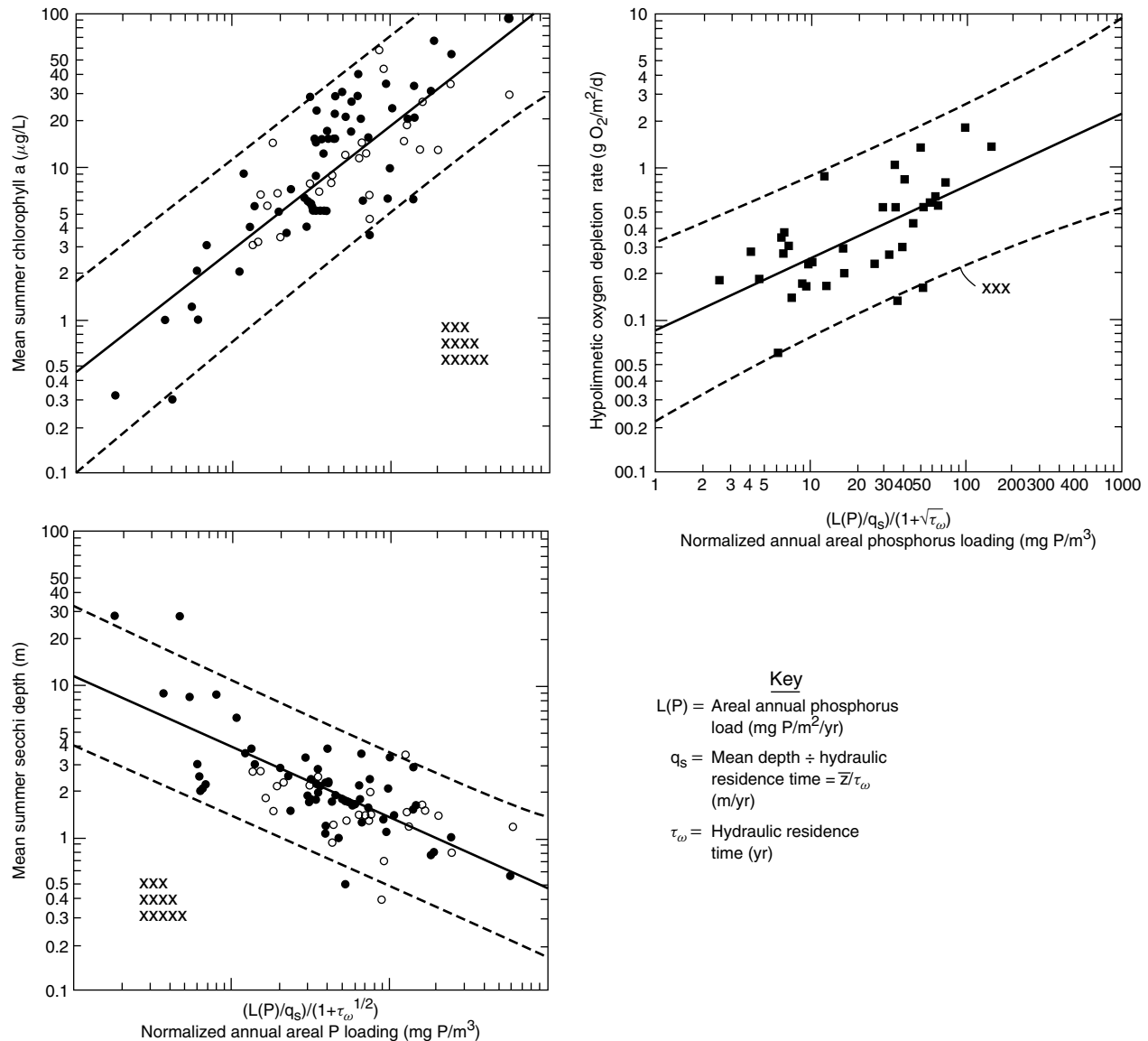


Figure 3. Relationships between normalized P load and eutrophication-related water quality response—U.S. OECD Eutrophication Study Results [after Rast and Lee (17)].

If the water quality problem is related to planktonic algae, the Vollenweider–OECD eutrophication modeling approach is the recommended approach for determining the reduction in nutrient loads/concentrations necessary to achieve the desired nutrient-related water quality in many lakes and reservoirs. Described by Rast and Lee (17) and amplified by Jones and Lee (18,19), this model empirically relates normalized phosphorus loading to eutrophication-related water quality parameters of chlorophyll, water clarity, and hypolimnetic oxygen depletion rate through relationships formulated by Vollenweider (20). These relationships take into account the influence of the key factors of the waterbody’s mean depth, hydraulic residence time, and surface area on the utilization of phosphorus by algae within a waterbody. These models, based on the OECD (21) and post-OECD Eutrophication Study data, are shown in Fig. 3. Each point in each figure represents a lake, reservoir, or estuary for which the nutrient load

and eutrophication response had been measured for at least a year to generate the model point. Jones and Lee (19) updated this model with data for more than 750 waterbodies in various parts of the world (Fig. 4). The use of this modeling approach and its reliability for predicting the changes in response parameters after a change in nutrient loading has been described by Rast et al. (22).

Rate of Recovery

One of the issues of particular concern in eutrophication management is the rate of recovery of a waterbody after reduction in the nutrient/phosphorus loads. Because large amounts of phosphorus are stored in lake sediments, some have incorrectly concluded that reducing the phosphorus load from the watershed would result in little improvement in water quality, especially in a waterbody with a long hydraulic residence time. However, Sonzogni et al. (23)

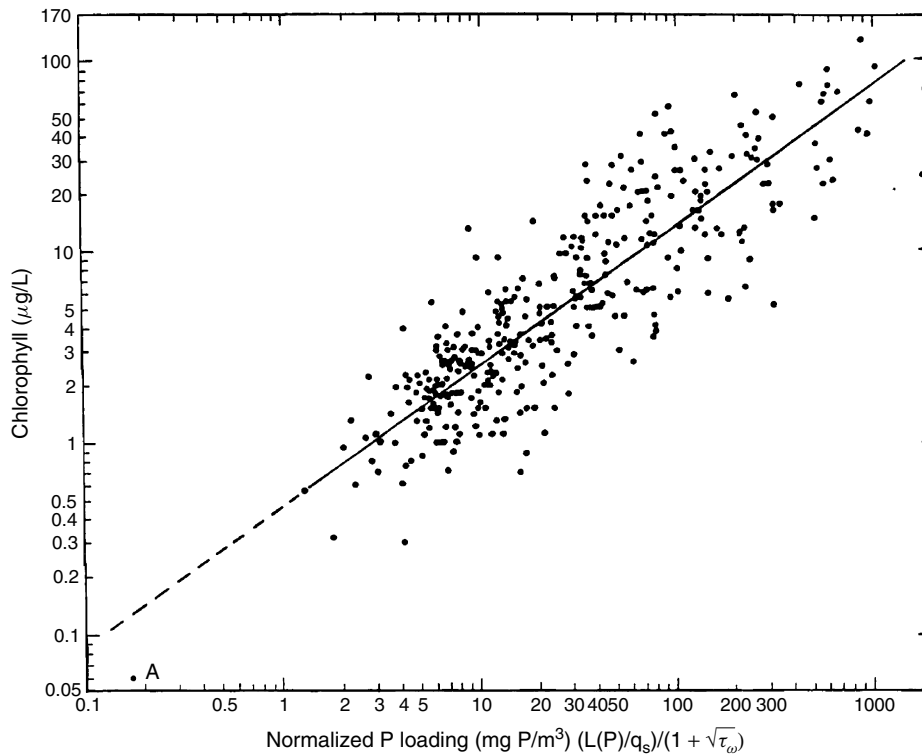


Figure 4. Updated relationship between normalized P load and planktonic algal chlorophyll response [after Jones and Lee (18)].

demonstrated that the rate of response in eutrophication-related water quality to reduction in phosphorus loading is governed by the phosphorus residence time in the waterbody. The P residence time in years is the total mass of phosphorus in the waterbody water column divided by the annual load, which is typically much shorter than the hydraulic residence time.

CONCLUSIONS AND RECOMMENDATIONS

Excessive fertilization, eutrophication, is a major cause of water quality impairment. Domestic wastewaters, urban stormwater runoff, and agricultural runoff/discharges are significant sources of nutrients that contribute to excessive fertilization of some waterbodies. Site-specific investigations are needed to determine the contribution of algal-available nutrients from these sources and the extent to which they can be controlled. Using the Vollenweider–OECD eutrophication modeling approach, the expected improvement in beneficial uses that could be achieved in many lakes or reservoirs by affecting a given load reduction and the expected recovery time can be estimated.

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CULTURAL EUTROPHICATION

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INTRODUCTION

The phrase “cultural eutrophication” (= cultural enrichment) is becoming widely used to denote organic pollution

resulting from human activities. Humans, through their various cultural activities, have greatly accelerated this process in thousands of lakes around the globe. Cultural or anthropogenic “eutrophication” is water pollution caused by excessive plant nutrients. Increased productivity in an aquatic system sometimes can be beneficial. Fish and other desirable species may grow faster, providing a welcome food source (1). Eutrophication produces “blooms” of algae or thick growths of aquatic plants stimulated by elevated phosphorus or nitrogen levels. There has been some uncertainty as to whether algal blooms result from increased concentrations of nitrate or phosphate or from some other cause. It is now commonly accepted that algal growth in fresh waters is generally restricted by phosphate concentrations, whereas in marine waters, it is restricted by nitrate concentration (2). In freshwaters, the nitrate concentration might, however, influence the kinds of algae that grow, some of which taint in drinking water or are toxic to animals.

Bacterial populations increase due to larger amounts of organic matter. The water becomes cloudy or turbid and has unpleasant tastes and odors. Cultural eutrophication can accelerate the “aging” of a waterbody enormously over natural rates. Lakes and reservoirs that normally might exist for hundreds or thousands of years can be filled in a matter of decades.

Cultural eutrophication also occurs in marine ecosystems, especially in near-shore waters and partially enclosed bays or estuaries. Partially enclosed seas, such as the Black, Baltic, and Mediterranean Seas, tend to be in especially critical condition (1). During the tourist season, the coastal population of the Mediterranean, for example, swells to 200 million people. Of the effluents from large cities, 85% go untreated into the sea.

SOURCES OF NUTRIENTS

Humans add excessive amounts of plant nutrients (primarily phosphorus, nitrogen, and carbon) to streams and lakes in various ways. Runoff from agricultural fields, field lots, urban lawns, and golf courses is one source of these nutrients. Untreated or partially treated domestic sewage is another major source. Sewage is a particular source of phosphorus to lakes when detergents contain large amounts of phosphates. The phosphates act as water softeners to improve cleaning action, but they also are powerful stimulants to algal growth when they are washed or flushed into lakes.

Agricultural Runoff

The enrichment material in agricultural runoff is derived from fertilizers applied to crops and from farm animal houses. Nitrogen used as fertilizer may be converted to nitric acid in soil and solubilize calcium, potassium and other ions, which become highly liable to leaching.

Domestic Sewage

Sewage is the most common source of nutrients and organic matter and undoubtedly, the greatest contributor to the eutrophication of lakes and ponds. Large quantities

of nitrogen and phosphorus excreted by humans and animals enter into sewage. Phosphatic detergents in sewage (without tertiary treatment) may contain 15 to 35 mg/L of total nitrogen and from 6–12 mg/L of phosphorus (3). Untreated sewage, besides nutrients, also adds large quantities of nitrogenous organic matter.

Industrial Wastes

The nutrients in industrial effluents are variable in quality and quantity, depending on the process and type of industry. The wastes from certain industries, particularly fertilizers, chemicals, and food, are rich in nitrogen and phosphorus. Organically held phosphorus is more soluble, and there is concern that it will leach into surface waters, giving concentrations of 1 mg/L or more, when large amounts of cattle and pig slurries are applied to sandy soil, as in the Netherlands (2).

Urban Runoff

Urban runoff contains storm water drainage with organic and inorganic debris from various paved and grassed surfaces and fertilizers from gardens and lawns.

EFFECTS OF CULTURAL EUTROPHICATION

1. The excessive growth, or "blooms," of algae promoted by phosphates changed the water quality in Lake Erie and many other lakes. These algal blooms led to oxygen depletion and resultant fish kills. Many native fish species disappeared and were replaced by species more resistant to the new conditions. Beaches and shorelines were fouled by masses of rotting, stinking algae.
2. Decomposition of algal bloom leads to oxygen depletion in water. This with a high CO₂ level and poor oxygen supply, aquatic organisms begin to die, and the clean water turns into a stinking drain.
3. Algae and diatoms attain a high degree of dominance due to overfertilization. Algae and rooted weeds interfere with hydroelectric power, clog filters, retard water flow, and affect water quality and water works.
4. Macrophytes, particularly *Hydrilla*, *Potamogeton*, *Ceratophyllum*, and *Myriophyllum*, assume high population densities and make near-shore and shallow regions unsuitable for any purpose.
5. Filamentous green algae, such as *Spirogyra*, *Cladophora*, and *Zygnema*, form a dense floating mat or "blanket" on the surface when the density of the bloom becomes sufficient to reduce the intensity of solar light below the surface. These blankets often give shelter to several undesirable insects, including mosquitoes.
6. Eutrophication of a moderate level may be beneficial to fish production as it increases the food supply for fish in the form of algae. Fish ponds are often fertilized with nutrients to accelerate algal growth and increase fish productivity. But because the level of eutrophication increases due to human activities,

the dominance of algal groups is taken over by blue-greens and the edible or game fish are replaced by hardy species of very little economic value.

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FISH CELLS IN THE TOXICOLOGICAL EVALUATION OF ENVIRONMENTAL CONTAMINANTS

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INTRODUCTION

Aquatic life at risk to modulation by environmental contaminants include fish and their organs, tissues, cells, and subcellular processes. Through branchial, dermal, and oral absorption, as well as biomagnification, it is highly probable that fish are exposed to many chemical species, their metabolites, and their mixtures (e.g., aromatic hydrocarbons, carbamates, heterocyclic compounds, heavy metals, organophosphates, and halogenated compounds). Sources of toxicant exposure include primary anthropogenic emissions, municipal and hazardous waste landfills, incinerators, episodic and diffuse loadings, as well as global secondary sources that involve complex cycling across air–water (e.g., deposition, rain, snow), sediment–water, and biotic interfaces (e.g., vegetation, fish, birds).

Environmental contaminants of concern require research into their accumulation potential, toxicological potency, and health effects, because they can adversely affect aspects of fish life histories through direct effects (e.g., on developing eggs and larvae) or by more indirect means (e.g., immunosuppression, and enhanced skin and liver disease). Chemical contaminant exposure can interfere with critical phases of the cellular response by destroying, sensitizing, or otherwise altering the functions of cells.

Fish models are increasingly recognized as useful for basic research, biomedical (e.g., carcinogenicity, pathology) and biotechnological applications (e.g., functional and comparative genomics, DNA isolation), and toxicity/environmental safety testing (e.g., lethality, endocrine and androgen modulation, developmental and immunotoxicity). Within these disciplines, fish cells are recognized as an important model because fish are among the oldest

and most diverse vertebrates known, are an ecologically critical and pervasive species, are abundant and widely distributed, play a major role in the energetics of trophic levels, have feasible and practical considerations as test organisms (e.g., small size, economical maintenance and use, fecundity, smaller volume of test chemical use and disposal), and often as top predators are at additional risk of bioaccumulating chemical contaminants.

Fish species that are increasingly being used in toxicity testing include the following: rainbow trout, lake trout, fathead minnow, bluegill, zebrafish, guppy, carp, barbels, mummichog, yellow perch, and medaka. Trends in microbiotesting and high volume/throughput testing will lead to further developments in cost-effective microscale aquatic toxicity testing that use fish cells. Fish cells are also becoming recognized as more relevant toxicologically; recent evidence suggests that bloodborne environmental contaminants may be more bioavailable for transport inside the cell, versus transport to circulating binding proteins, than previously suspected. Surrogate tissues such as blood may also serve as an indication of target tissue exposure and cellular stress. The use of fish cells also enables investigations that seek to characterize the mechanisms of toxic effects.

Trends in Toxicity Testing

Prominent trends in toxicity testing include more *in vitro* tests and especially *in vitro* mechanistic assays, aimed at appreciating the importance of the molecular and cellular bases for the effects of chemical toxicants. For example, the assessment and role of cellular apoptosis in immunotoxicological methods is gaining recognition. Characterizing xenobiotic-induced or inhibited apoptosis provides the opportunity to detect subtle and reversible changes in the normal activity of cells and aids in explaining fish stress and time from exposure to toxicity. Other popular end points include heat shock proteins, signal transduction, endocrine modulation, oxidative stress, enzyme induction, influence of inflammatory mediators and endotoxin exposure, and cholinesterase activity. Further, cells harvested from hepatocytes and gills remain prominent choices for toxicity testing. These toxicological end points at biological and biochemical cellular levels attempt to evaluate the functional competence of specific organ systems and individuals.

In vitro response tests are often selected for the following reasons: primary cell exposures provide the necessary dose-response and mechanistic toxicity data; *in vitro* tests require minimal sacrifice of whole fish and minimal use of chemicals; tests are subject to less ethical scrutiny when using cells versus whole animals; and *in vivo* exposure response tests are often impractical because the fish are collected some distance from the laboratory and are only available infrequently and seasonally. The measured end points for assays with fish cells lines often include detection by flow cytometry and fluorescence microscopy. With regard to fish systems, *in vitro* immune cell toxicity studies have focused on the reduction of viability (e.g., assays such as trypan blue, neutral red differential uptake) or reactivity (e.g., phagocytic indexes).

Biomarkers are increasingly employed to evaluate chemically induced alterations in fish at the molecular, biochemical, cellular, or organismal level. These techniques that detect cellular departures from homeostasis may be applied in experimental effect studies or field monitoring and may be destructive or nondestructive to the fish test organisms. However, many of the classical biomarkers of exposure such as enzyme induction and tissue residue fail to identify toxicological hazards and clinical implications adequately that can impact exposed individuals and populations.

In all cases, good laboratory practice and appropriate methods for analgesia, anesthesia, and euthanasia must be employed. Standard operating procedures for tissue harvesting and cell isolation must be designed to minimize handling time prior to preservation or assay testing.

Importance and Policy Relevance

Fish, such as the lake trout in the Great Lakes, can be designated sentinel species of ecosystem health and are useful indicators for assessing chemical contaminant stress. Fish cells in toxicology research can delineate the magnitude of adverse effects of environmental contaminants and one can also extend the relevance of the data to understanding *in situ*, *in vivo*, and ecosystem end points.

With regard to assessing aquatic ecosystem health, toxicological end points at the cellular level can attempt to evaluate the structural integrity and functional competency of specific organ systems and individuals and attempt to address both quantitatively (e.g., internal and biologically effective dose) and qualitatively (e.g., early biological effect, altered structure/function) the more subtle toxic effects of low level contaminant exposure. The fish cell toxicity end point is important because it is rarely possible to predict the effects of urban, industrial, and agricultural pollutants on aquatic biota based solely on the composition and concentrations of contaminants. Thus, employing fish cells in the toxicological evaluation of environmental contaminants is important because it provides a potentially valuable risk assessment component, as well as tools for policy-makers and environmental, epidemiological, and fishery scientists.

Although the fish cells are important models of investigation independently, there is comparative evidence that fish cells and those of higher vertebrates are functionally similar. Further, ambient exposure routes for fish may approximate chronic inhalation tests in mammalian and vertebrate models. However, there are important differences between fish and human cells (e.g., the greater membrane rigidity, volume, surface area, and presence of a large nucleus in fish red blood cells).

In addition, there are potential challenges and obstacles in the interpretation and extrapolation of *in vitro* studies of fish, including (1) relevance of *in vitro* dose levels employed in the laboratory compared to *in vivo* environmental exposures; (2) ability to rule out concurrent exposure to other chemical, biological, or physical stressors; and (3) ascertaining clinical/environmental significance based on statistical significance.

Nonetheless, *in vitro* investigations of this type have advantages over *in vivo* and field studies for the following reasons: control over environmental exposure conditions, potentially reduced variability between experiments, the ability to evaluate thousands of individual cells readily, and the requirement of a small quantity of test chemicals to complete the exposure–response studies.

From an institutional viewpoint, some environmental policies state that contaminant-induced cellular changes are to be considered in decision making because they may directly influence ecologically important parameters, as well as human welfare. The science as applied to fish cells may not be sufficiently developed to allow rigorous hazard assessments, but there remains growing scientific and policy understanding of the subtle yet adverse effects of persistent contaminants on fish physiology.

Bioavailability and Mode/Mechanism of Action

In the aquatic environment, it is important to distinguish between the forms of toxicant exposure especially with regard to bioactivity, biodistribution, and potential toxicity. Speciation and bioavailability are strongly dependent on environmental factors (e.g., pH, redox state, dissolved oxygen, humic content, selenide and sulfide levels, mineral content, mercury content), physicochemical factors (e.g., solubility, partitioning, metal–ligand complexing, ionization), and biological factors (e.g., presence of methylating or demethylating microbes). In teleost species, some of the bioavailability is determined by the pH and chloride concentration of the stomach; at low pH and high chloride concentration, stomach conditions favor the formation of species more readily transported into the blood stream. There are also a number of intrinsic and extrinsic factors that may interact with toxic end-point parameters under investigation (e.g., reproductive status, age, stress, nutrition, toxicant mixtures, previous exposure, predation, food chain effects, habitat, density, environmental stochasticity, concurrent infections, stress, species, and genotype).

Knowledge of chemiometabolic enzyme and receptor systems in fish may prove useful in improving the interpretation of toxicological studies. Fish can clear select chemical toxicants via metabolic routes to a minor degree (e.g., feces, extraction from gill membranes, urine, bile, eggs, and mucus). It should be noted that in fish, as in mammals, the most important enzymes of chemical biotransformation include cytochrome P450, UDP-glucuron-, glutathione-, and sulfotransferases; and they are found most appreciably in the liver, although activity is found in the gills, intestines, and kidney. Interestingly, research evidence supports the presence of the aryl-hydrocarbon (Ah) receptor in teleost fish.

Characterizing the mechanisms of action of chemical toxicants is important for physiological, toxicological, and therapeutic reasons. For instance, once incorporated into biological tissues, the physiological and toxicological effects of metals are regulated by binding to specific ligands and excretion, involving competition for transport and cellular sites and inhibition of enzyme systems. Chemically induced toxicity may involve cell volume changes, alterations in cell permeability, and phospholipid structure. These chemically induced changes may lead to suppressed

cell competence, induced or inhibited normal cell proliferation, altered hematology, altered ion balance and metabolism, and enhanced susceptibility to disease states. Certainly mechanistic investigations in ecotoxicology and etiopathogenesis have gained increasing importance especially with regard to prophylactic measures.

Uncertainty Analysis

By definition, uncertainties are inherent in ecological risk assessments and must be addressed in toxicological evaluations using fish cells. The sources of uncertainty include measurement or estimation of variables, natural variability/environmental stochastic over time and space (e.g., rainfall, wind velocity, temperature), and use of models that do not accurately reflect the environment or exposed populations of concern.

During toxicity testing, there is potential for notable individual variations in sensitivity and response. Certainly, it should be noted that overall *in vitro* toxic effects on fish do not necessarily also lead to *in vivo* toxicity. Many *in vitro* systems lack cell-to-cell contacts that may make them behave differently, as well as the influence of regulation by other physiological systems (e.g., nervous or immune). *In vitro* toxicology and mechanistic models are only as good as the level of *in vivo* understanding, and both must be considered in characterizing a chemical hazard. Further, subtle perturbations in cell function following *in vitro* exposure may not in every instance result in a relevant clinical effect, especially given the functional reserve, complexity, and adaptive responses. Other considerations for explaining differences in sensitivity can be attributed to genetics (e.g., expression of metallothionein), metabolism (although Phase I and II metabolism is similar to that of mammals), DNA repair, as well as the varying complexity of teleost fish tissue (e.g., for the immune system, the absence of lymph nodes, tonsils, bone marrow, and the presence of the pronephros).

Other sources of uncertainty are inherent and include experimental design and conduct, variability in chemical composition and purity, variability between sexes, variability across experiments, experimental uncertainty, animal to human extrapolation, higher dose to lower dose extrapolation, difference between commercial and environmental chemical species, persistence and exposure duration, and human variability in exposure and sensitivity. Finally, there is a fundamental assumption in toxicity testing that may lead to uncertainty—that lab tests of single individuals and relatively constant exposures are predictive of field-based intermittent exposures.

SUMMARY

In the end, the significance of studying fish cells in environmental toxicology and hazard assessment research is twofold: *first*, it provides comparative results of a battery of potential *in vitro* assays as an alternative system to *in vivo* toxicity tests and for screening and evaluating chemicals of concern; and *second*, it provides methods for diagnosing and predicting modulation and toxicity from low level exposure to environmental

chemical agents that will aid in defining the mechanisms responsible for the observed effects. More investigations into the cellular responses of fish when given equally relevant and equitoxic doses are an important avenue for further study. A battery of available toxicity test methods on fish and fish cells will remain a backbone of environmental safety testing and chemical/biological risk analyses, especially given economic, logistical, and ethical concerns.

FISH CONSUMPTION ADVISORIES

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INTRODUCTION

Fish consumption advisories can be defined as government health advice issued to the public concerning deleterious substances in fish and/or shellfish (e.g., crabs). These advisories typically provide advice on avoiding or limiting the consumption of certain types or sizes of fish, or on the number of fish meals to consume over a period of time in order to limit health risks. Fish consumption advisories were first issued in the United States in the mid-1970s (1).

HISTORY

Occasional contamination of shellfish and, to a lesser extent, finfish with pathogens and parasites has long been recognized as a potential public health hazard. Prior to fish consumption advisories addressing risks from environmental chemical contaminants, biological (e.g., bacterial) contamination resulted in the closure of shellfish beds and the removal of contaminated harvests from commerce. Such actions tend to be legally enforceable and function through the exercise of public health law. As such, they are not technically advisories, which, by their nature, are provided as guidance and rely on voluntary adherence. Nonetheless, experience with such actions paved the way for governments to consider fish consumption advisories as part of their larger public health responsibility. The recognition of environmental mercury poisoning through fish consumption in the Minimata Bay area of Japan in the 1950s raised the public consciousness about the potential for anthropogenic chemical contamination of dietary fish. However, at that time, and for several decades afterwards, concerns about mercury in fish focused on frank poisoning resulting from highly elevated levels. In addition, the impact of the Minimata contamination on an entire community consuming a common source of fish tended to focus attention on commercial fish. The indirect sources of mercury contamination of otherwise pristine waterbodies and its impact on recreational and

subsistence anglers was not fully recognized until the early 1990s. Thus, mercury contamination in fish was treated similarly to biological contamination, and was addressed through regulatory control of permissible mercury levels in commercial fish. By the mid-1990s, recognition that mercury in fish posed health risks more subtle than frank poisoning led to the adoption of consumption advice to recreational and subsistence anglers based on maintaining exposure below a "virtual" safe level and, at the same time, advising fish consumption up to the maximum safe exposure level. The intent of this approach is to maximize the nutritional benefits to be derived from safe levels of fish consumption while avoiding the health risks associated with contaminant exposure. More recently, this approach has been applied to commercial fish, with the federal government (i.e., U.S. Food and Drug Administration, FDA) and various state governments issuing consumption advice for fish whose contaminant levels are nonetheless within regulatory limits for sale.

In general, the first major fish consumption advisories were issued for PCBs (polychlorinated biphenyls). In 1973, the FDA established a tolerance (i.e., a regulatory limit) of 5 parts-per-million (ppm) PCBs for fish in interstate commerce. The FDA subsequently lowered this tolerance to 2 ppm in 1984. Initially, many states used FDA action levels or tolerances in setting fish consumption advisories. However, unlike mercury, PCBs were recognized as carcinogens. Historically, noncarcinogens are considered to have a threshold level of exposure, below which no adverse effects are anticipated, whereas carcinogens have been considered to pose some risk at any level of exposure. Thus, rather than the all-or-nothing regulatory approach initially adopted for mercury, the presumption of some risk at all levels of consumption raised the need for advice that attempted to balance low risk against continued consumption. The FDA's tolerance limit was not established with such a goal in mind, and thus, starting in the late 1980s and early 1990s, some states began issuing consumption advice to recreational and subsistence anglers that was designed to keep lifetime cancer risk from consuming PCBs in fish within an "acceptable" range. Based on historical concepts of "acceptable" cancer risk, this range generally extended from a risk of one-in-a-million (1×10^{-6}) to one-in-ten-thousand (1×10^{-4}) of developing a PCB-attributable cancer over the course of a 70-year lifetime. This approach was subsequently applied to known or suspected carcinogenic pesticides (e.g., chlordane).

SCOPE

Advisories are issued by local, tribal, state, and national (e.g., U.S. EPA) agencies. Typically, most advisories are issued by the state health department or environmental/natural resource agency. Based on data compiled by the U.S. EPA (2), 48 states had issued a total of 3,089 advisories in 2003, including species-specific, water body-specific, and statewide advisories. These advisories covered 35% of the nation's total lake acres and 24% of the nation's total river miles, and included coastal waters. Together, mercury and PCBs account for the great

majority of advisories, with chlordane, dioxin, and DDT accounting for nearly all of the remaining advisories (2). A listing of all state advisories can be found on the U.S. EPA website (<http://www.epa.gov/ost/fish/>). In 2004, the U.S. FDA together with the U.S. EPA (3) issued a national mercury advisory for commercial and recreationally caught fish. This advisory included a “do not eat” advisory for shark, swordfish, king mackerel, or tilefish for the sensitive population (women of childbearing age, pregnant and breastfeeding women, and children). It also advised this sensitive population to check local fish advisories, or if no advice is available, to limit intake to one (6 ounce) meal per week of locally caught fish. Australia and New Zealand have offered similar advice for higher trophic level commercial species (e.g., marlin, shark, swordfish). The European Food Safety Authority (4) has also issued a mercury advisory to this sensitive subpopulation in the European Union concerning consumption of fish. Member countries in the European Union, other countries, and regional governments also issue fish advisories (e.g., British Food Standards, Health Canada, Ontario Ministry of the Environment) (5).

THEORY

Fish consumption advisories seek voluntary reduction of risk through modification of fish consumption behavior. Regulatory restrictions are generally applied when fish enter into commerce or when the risk posed by consumption is clear and immediate. In contrast, advisories tend to be predicated on the notion that government has, at most, limited authority to regulate individual behaviors like fish consumption. At the same time, government operates under a mandate to protect the public health, which is generally seen as a responsibility to advise, inform, and educate. Nonetheless, occasions exist where health risks are imminent or sufficiently elevated as to justify regulations closing specific waterbodies to fishing for some or all species. Often no clear *a priori* dividing line exists between conditions requiring advisory and regulatory actions, and decisions are often made on a case-by-case basis. Advisories are most appropriate when the risk is low to moderate, and are potentially balanced by associated benefits such as nutrition and the continuation of cultural practices. The most basic type of advisory is dichotomous—eat/don’t eat. This approach is simple to communicate, but offers little opportunity to consider benefits as well as risks. More sophisticated advisories provide consumption frequency advice (e.g., no restriction, eat once per week, eat once per month... don’t eat). Which frequency of consumption is recommended depends on the concentration of the contaminant in the particular species of fish such that eating a typical size serving with the specified frequency will not result in exceeding the maximum acceptable risk level. Thus, a fish for which an eat-once-per week advisory is issued would generally have about 25% of the concentration of the same contaminant as a fish for which an eat-once-per month advisory is issued. Consumption frequency approaches allow consumers to eat the maximum amount of fish that is consistent with remaining within a safe level of exposure

or an “acceptable” level of risk. As the contaminant level in any given fish is not known, advisories are derived by extrapolating contaminant data from a limited sample of a given species of fish from a specific water body. Sometimes regional or statewide advisories are constructed by generalizing sample data from a group of representative waterbodies. Various statistical approaches are used for this extrapolation, including averaging concentration and selecting a concentration representing a given percentile of the distribution of sampled fish.

Advisories are generally constructed using common, default assumptions for factors such as body weight and portion size. For contaminants to which one subgroup in the population is more sensitive than the general population (such as methylmercury, which effects fetal neurological development at levels below those causing neurotoxicity in adults), advisories may provide separate advice to the sensitive group and the general population, which includes using different default body weights appropriate to each group

METHODS

Sampling

Fish tissue contaminant data are needed in order to develop fish advisories. Sampling methods vary by region and target species. Collection methods include traps, gill nets, trawls, electrofishing, hook and line fishing, as well as purchase from recreational or commercial fishermen. Fillet samples are typically analyzed, as these are often the target tissue for consumption, but some cultures do cook whole fish. Specialized tissues can also be targeted for certain species (e.g., hepatopancreas or “green gland” for crabs and lobster) in addition to the muscle. Individual fish samples or composite samples (i.e., multiple fish per sample) can be collected depending on needs and budget. Information on monitoring strategy, field procedures, target species, and target analytes are detailed by the U.S. EPA (6).

Analysis

A variety of methods are used to measure contaminants in fish species. Analytical methods and instrumentation have improved dramatically over the past few decades, allowing lower detection limits and compound discrimination (e.g., PCB congeners). For mercury, the cold vapor atomic absorption spectrophotometry is typically used (7). For organic contaminants (e.g., PCBs, pesticides, dioxins/furans, and polycyclic aromatic hydrocarbons), the methods vary, but gas chromatography/electron capture detection (GC/ECD) or GC/mass spectrometry (GC/MS) techniques are normally used (7). Additional information on sample handling, analytical methods, quality assurance, quality control, and data analysis are available (e.g., 6).

Risk Assessment

Risk-based advisories may be based on carcinogenic effects or noncarcinogenic effects. A toxicity factor (slope factor

or reference dose, as described below) is based on toxicity data, usually from experimental animals and, less often, from humans. Advisories based on carcinogenic effects generally assume that no threshold exists below which there is no risk, so that any exposure to the contaminant poses some risk of cancer. Therefore, a target lifetime risk level (typically 10^{-4} to 10^{-6} , or 1 in 10,000 to 1 in 1,000,000) is chosen based on policy rather than scientific considerations. A carcinogenic potency factor (also called a slope factor) that relates daily dose to risk is developed from the toxicity data and is used to determine the daily dose of the contaminant (7).

$$\text{Daily Dose (mg/kg body weight/day)} = \frac{\text{Lifetime Risk (unitless)}}{\text{Potency factor (mg/kg/day)}^{-1}} \quad (1)$$

Advisories based on noncarcinogenic effects use the assumption that a threshold exists below which adverse effects other than cancer, such as organ toxicity, developmental effects, or reproductive effects are unlikely. A reference dose (in units of mg/kg body weight/day), below which no adverse effects are expected in the overall population, is developed by applying appropriate uncertainty factors to the dose at which effects occur in animals or humans, which is the daily dose that generally forms the basis for risk-based consumption advisories for noncarcinogenic effects.

Consumption advisories for fish with a given contaminant concentration can be derived as follows:

$$\text{Consumption Rate (meals/day)} = \frac{\text{Daily Dose (mg/kg/day, see above)} \times \text{Body Weight (kg)}}{\text{Contaminant Concentration (mg/kg)} \times \text{Meal Size (kg/meal)}} \quad (2)$$

Assumptions must be made for meal size (typically 8 ounces or 227 grams) and body weight (typically 70 kg for an adult male and 62 kg for a pregnant female). Equation 2 provides the consumption rate in meals/day. If the consumption rate is calculated to be sufficiently large (e.g., 1 meal/day or greater), the advisory can be given as “unlimited consumption.” If the resulting consumption rate is less than the frequency corresponding to “unlimited consumption,” the advisory can be expressed in a convenient unit, such as meals/week, meals/month, meals/3 months, or meals/year. If the consumption rate is less frequent than a reasonable minimum (e.g., less frequent than once per year), the advisory may be given as “Do not eat.” The equations given above are a generalized form of the approach given by the EPA (6), which provides separate equations for carcinogens and noncarcinogens.

Separate fish consumption advisories may be developed for a contaminant to protect the general population and sensitive subpopulations. For example, for methyl mercury and polychlorinated biphenyls (PCBs), neurologic development is the endpoint of concern in the developing fetus, nursing infant, and young child. For the general

population, paraesthesia (tingling of the extremities) is the endpoint of concern for methyl mercury and cancer is the endpoint of concern for PCBs.

Fish consumption advisories that do not rely directly on the risk-based approach described above have also been issued. For example, a general advisory has been issued by some states (e.g., Pennsylvania), advising consumption no more than one meal per week of freshwater sport fish. This approach is based on the precautionary principal that can be stated as: in the absence of specific data and given the tendency for bioaccumulative contaminants to be present in freshwater fish, it is prudent to limit consumption.

Another possible approach, for chemicals for which background dietary exposure from sources other than fish results in considerable risk, is to base the advisory on permitting a fractional increase above the background level. For example, background exposure to dioxin, primarily from dietary sources including meat and dairy product, have been estimated to result in a lifetime cancer risk of 1 in 1000 or 10^{-3} (8). This risk level is far above the risks typically used as the basis for risk-based consumption advisories. In such a case, the risk-based advisory approach would limit the consumption of fish, which is a beneficial part of a healthy diet, without actually providing any substantial reduction in risk from dioxin and related compounds.

ADVISORY EXAMPLE

Local or state advisories are typically more specific in terms of fish size, water body, and/or population as compared with national advisories. For example, in New Jersey, mercury advisories for largemouth bass list an advisory of “one meal per month” in Newton Lake for the general population and “do not eat” in Cooper River Lake for children and women of childbearing age (Table 1). Whereas, dioxin (2,3,7,8-tetrachlorodibenzo-*p*-dioxin) advisories in the Passaic River, NJ are “do not eat” for all fish species for all consumers (9).

COMMUNICATION

A very important aspect of fish consumption advisories is outreach to the public. Advisories result in public health protection only if their message is received, understood, and acted on. Outreach activities involve identifying populations at risk, developing an effective communication strategy, and implementation of that strategy using a variety of methods. For example, government agencies have used a variety of techniques, both traditional and unique, to get the advisory message to the public. Examples include publications (e.g., brochures), listing of advisories with fish regulations, posting on web pages, issuing press releases, public service announcements, videos, posting of warning signs at boat ramps and other public access points, direct outreach efforts using local community groups, and toll-free numbers. Risk communication guidance is available from government sources (e.g., 1).

Table 1. Example of Fish Consumption Advisory (8)

Location	Species	Advisory/Prohibition		
		General Population ^{a,b} Range of Recommended Meal Frequency		High-Risk Individual ^{b,c}
		Lifetime Cancer Risk of 1 in 10,000	Lifetime Cancer Risk of 1 in 100,000	Recommended Meal Frequency
		Do Not Eat More Than	Do Not Eat More Than	Do Not Eat More Than
Cooper River Lake (Camden Co.)	Largemouth Bass	Four meals per year	Do not eat	Do not eat
	Common Carp			
	Brown Bullhead	One meal per week	One meal per month	One meal per month
	Bluegill Sunfish			
Newton Lake (Camden Co.)	Bluegill Sunfish	One meal per week	One meal per month	One meal per month
	Brown Bullhead			
	Largemouth Bass	One meal per month	Four meals per year	Four meals per year
	Common Carp		One meal per year	Do not eat
Passaic River downstream of Dundee Dam and streams that feed into this section of the river.	All fish and shellfish*	Do not eat		Do not eat

^a Range of Recommended Meal Frequency corresponds to a cancer risk of 1 in 10,000 to 1 in 100,000 over a lifetime.

^b Eat only the fillet portions of the fish. Use proper trimming techniques to remove fat, and cooking methods that allow juices to drain from the fish (e.g., baking, broiling, frying, grilling, and steaming). One meal is defined as an eight-ounce serving.

^c High-risk individuals include infants, children, pregnant women, nursing mothers, and women of childbearing age.

*Selling any of these species from designated waterbodies is prohibited in New Jersey (N.J.A.C. 7:25-18A.4).

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FISHERIES: HISTORY, SCIENCE, AND MANAGEMENT

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The overall goal of fisheries management is to produce sustainable biological, social, and economic benefits from renewable aquatic resources. Fisheries are classified as *renewable* because the organisms of interest (fish, shellfish, reptiles, amphibians, and marine mammals) usually produce an annual biological surplus that, with judicious management, can be harvested without reducing future productivity. In contrast, *nonrenewable* resources

(oil, coal, iron, and copper) are available in fixed quantities and are not replaced except over geologic time.

The benefits that humans gain from a fishery are diverse and may be enumerated in several ways. Most commonly, benefits are computed as *commodity* output—the weight or number of fish produced. Commodity output may be further split between the animals harvested by capture (fishing for wild animals) or culture (produced as captive animals)—commonly called the *capture* fisheries and the *culture* fisheries, respectively.

FISHERIES MANAGEMENT

The benefits are commonly measured as wholesale or retail economic *value* of the commodity output. Such benefits are easily calculated for *commercial* fisheries because the products are usually sold, but for sport or *recreational* fisheries, the quality of the fishing experience is very important, so measures of catch in weight, number, or value only partially measure the benefits provided to fishermen or to society. Measurements of the indirect economic value of recreational fishing that include the quality of the fishing *experience*, however, remain controversial. Even in commercial or subsistence fisheries, substantial benefits may be associated with *cultural* or *religious* aspects. Although such benefits are difficult to measure, they may be very important to the participants.

Beyond the direct benefits derived from harvested fish or the fishing experience, benefits are also derived by individuals and society from simply *knowing* that a particular natural resource exists (often called *existence* value). Society and individuals receive intangible benefits from preserving species and habitats, especially those in danger of extinction. Such benefits are often significant, but, like the benefits from recreational fishing, they are also exceedingly difficult to quantify in economic terms. The whale fishery is an example for which the value of leaving the animals unharvested currently is of greater benefit (primarily intangible) in most societies than the value (economic) of the harvested animals.

Whether or not measurable, fisheries management is increasingly being guided by ecological benefits mandated in treaties, laws, and government policies. For example, the *Convention on Biological Diversity* obligates signatory nations to preserve their biological diversity to the maximum possible extent. Many nations also have laws to protect species at risk of extinction, and these laws may be important constraints on the scope, type, and intensity of fishing that will be permitted.

In practice, the overarching management policy goal for managing a nation's fisheries is often stated in general terms such as:

To ensure the attainment and continued satisfaction of human needs for present and future generations in an environmentally non-degrading, technically appropriate, economically viable, and socially acceptable manner, and such that land, water, plant, animal, and genetic resources are maintained.

The challenge for the fisheries manager is to translate such a general policy goal into a practical, effective program to maximize the benefits of specific fisheries to society.

FISH VERSUS FISHERIES

The words “fish” and “fisheries” have several meanings, and these terms often cause confusion. As traditionally used in fisheries management, *fish* typically includes the entire suite of *aquatic* organisms that are harvested (mackerel, tilapia, tuna, guppies, sea turtles, seals, whales, sea urchins, clams, squid, and frogs), or *could* be harvested if their numbers permitted. Thus, the term *fish* is not solely the *fin fish* (fish that have fins), so a fisheries manager may work with turtles, squid, or sponges, rather than fin fish. *Shellfish* (clams, crabs, lobsters) are also included under the broad definition of *fish*. In contrast to *fisheries* managers, *wildlife* managers generally deal with terrestrial mammals and birds (deer, wolves, bears, ducks, hawks, and whooping cranes).

A *fishery* is defined generically as a system composed of three interacting components: the aquatic biota, the aquatic habitat, and the human users of these renewable natural resources. Each of these components influences the fishery's performance. Understanding the entire system and its parts is often essential to successful management of a fishery.

There are many different types of fisheries and they may be classified in several ways:

- **Type of environment** (freshwater habitats—lakes, reservoirs, rivers, streams, and ponds; saltwater habitats—estuarine, coastal, and open ocean).
- **Method of harvest** (seining, trolling, trawling, fly casting, spearing, and dip netting).
- **Type of access permitted** (open access to fishing, open access with regulation, limited or purchased access, private property).
- **Organism of concern** (salmon, shrimp, bass, turtles, squid, cod, sharks, sea horses, whales, and swordfish).
- **Purpose of fishing** (commercial fishing for a product to sell, subsistence fishing for direct food consumption, or recreational fishing for sport and leisure).
- **Degree of wildness** of the target animals (totally wild and free-roaming animals, totally captive animals grown in ponds, or animals spawned in captivity, but released in the wild to be captured when they mature).

HISTORY OF THE HUMAN/FISH RELATIONSHIP

Fish have occupied an important place in human society for thousands of years. Early humans obtained fin fish, shellfish, and other aquatic life along the shores of lakes, rivers, and oceans. Archeological records document the use of fish spears 90,000 B.P. (90,000 years before present), nets 40,000 B.P. and fish hooks 35,000 B.P. The earliest documented human communities dependent on fishing were in the vicinity of Lake Mungo (Australia) 30,000 B.P. and Crete 8,000 B.P. The Egyptian aristocracy fished as a leisure activity at least 4,000 B.P. Fish have been raised in captivity for several thousand years.

As better preservation techniques developed (drying, smoking, and salting) and transportation improved,

commercial fishing in the Middle Ages began shifting from local, small-scale activities to commercial, large-scale enterprises. Boat design and construction advanced, along with corresponding improvements in fishing gear and preservation techniques, especially the advent of canning. Canning represented a particularly important advancement because it permitted long-term storage and large-scale distribution of fishery products.

Cod fishing off eastern North America began in earnest in the early 1500s. By the 1600s, whaling was a prominent activity in many high seas locations. In the late 1800s, steam-powered ships, along with mechanized fishing techniques and refrigeration, enabled development of the large-scale industrial fisheries that still exist today. During the past 100 years, the global level of fishing has expanded continuously, a trend disrupted only briefly during the two world wars. After World War II, the intensity of commercial fishing especially increased.

Today, commercial fishing continues as a major economic sector in many countries. In addition to the large worldwide value of the catch, approximately 36 million people (15 million full-time, 13 million part-time, and 8 million occasional) are employed in the capture and culture fisheries.

Although documented for thousands of years, *recreational* fishing only relatively recently has become an activity enjoyed by large numbers of people. The sport surfaced during the Renaissance as a socially acceptable leisure activity and received broad visibility from the 1653 publication of *The Compleat Angler* by Izaak Walton. By the mid-1800s, recreational fishing was an important and common activity, particularly in North America and Europe. The scale of recreational fishing expanded greatly after World War II, especially in North America.

Millions of people fish recreationally, and they support a multibillion-dollar sport fishing industry worldwide (fishing gear and equipment, bait, boats, motors, outdoor clothing, lodging, and food). Recreational angling ranges in extremes from the serenity of fly-fishing in a remote alpine stream, to the noise and excitement of a bass fishing tournament held in a large reservoir, to the excitement of a child catching its first fish from the bank of a pond.

Beyond the widespread pursuit of wild fish for commercial or recreational purposes, many aquatic species are now successfully raised in aquaculture facilities. Raising fish in captivity (especially various carp species) for food has been practiced in China for at least 4,000 years. The Chinese also developed effective means to breed and raise fish for ornamental purposes, an early precursor to today's vast aquarium market. By the mid-1900s, carp, tilapia, catfish, trout, salmon, and shrimp were widely raised for the *food* market; several of these and other species such as pike, sunfish, bass, and walleye were raised for *stocking* to enhance recreational fishing opportunities.

CURRENT USES OF FISHERIES RESOURCES

More than 4,000 species of aquatic animals are harvested worldwide, totaling approximately 120 million metric tons annually. The total harvest includes *capture* fisheries

(fish caught by nets, trawls, hooks) and *culture* fisheries (fish grown in ponds, cages, hatcheries), generally called *aquaculture*. The harvest tonnage from *capture* fisheries quadrupled between 1950 and 1990, but has leveled off or even declined since 1990. Harvest from *aquaculture* continues to increase. Fisheries harvest data are often of questionable accuracy, but the most recently available data on the combined capture and culture harvest suggest that China is the world's leading fish producer (32.5%), followed by Japan (5.1%), India (4.5%), United States (4.4%), and Russia (3.9%).

Aquacultural production continues to increase in importance worldwide and now accounts for approximately one-fifth of the total fish produced. Atlantic salmon, cultured in cages and pens, are raised in Norway, Scotland, Chile, Canada, United States, and elsewhere and provide fresh fish to the retail market year-round. Catfish, grown in ponds in the southern United States, and trout, grown in hatcheries in the northern United States, provide high quality, reliable, year-round products to the national retail market. Carp and tilapia are produced in large numbers, especially in Asia and Africa, and provide the primary source of animal protein for humans in those areas.

Worldwide annual per capita consumption of fish and shellfish is approximately 34 pounds (15 kg), but this varies considerably among regions and individual countries. Per capita consumption of fish has nearly doubled since the 1960s. Europeans and Asians tend to have the highest per capita consumption. Annual per capita consumption in the United States is approximately 15 pounds (7 kg); tuna, shrimp, pollock, salmon, and catfish are the top five.

In North America, the economic vitality of some rural communities depends on catering to the needs of recreational fishermen (fishing equipment, outdoor clothing, boats, motors, trailers, food, and lodging). In 1996, the estimated 35 million American adults (age 16 and older) who fished recreationally spent more than US\$37 billion for goods and services related to the sport. The indirect economic impact of this direct expenditure totaled more than US\$108 billion.

The market for aquarium and ornamental fish is important worldwide and continues to increase. Fish to supply this market are obtained from both wild and captive stocks. The live fish department is one of the most popular and profitable in many pet stores. Accompanying the sale of live fish is a market for aquaria, aquarium supplies, and, of course, fish food and medicine. Approximately 10 million ornamental *saltwater* fish are imported annually throughout the world. The number of *freshwater* fish imported is much higher and includes more than 5,000 species worldwide. The value of the U.S. retail ornamental fish market (live animals for aquaria and ornamental ponds only) is approximately US\$3 billion.

CHARACTERISTICS OF AQUATIC ENVIRONMENTS

Lakes, streams, ponds, oceans, and estuaries are biologically productive, but their productivity has ecological constraints, and these constraints can be reduced by human actions. Many aquatic habitats have been altered

by human actions (dredging, filling, damming, road building, pollution, and introduction of nonindigenous species), and their potential for producing sustained fish harvests has been reduced.

Variability is a pervasive characteristic of all ecosystems. Even in the total absence of human activity, aquatic ecosystems exhibit considerable fluctuations in the abundance of individual species. Fish species may vary in abundance several fold between years. In some years, there may be tremendous spawning success. In other years, the same species may have little or no reproduction. Variability in the aquatic environment makes it challenging to assess the likely biological consequences of fisheries management options such as adjusting harvest levels, changing gear regulations, or even placing a moratorium on fishing.

Climatic changes also alter the productive capacity of aquatic environments over the long term. Subtle shifts in ocean currents may cause some fish populations to collapse and others to thrive. Some changes in fish abundance caused by climatic or ocean shifts may happen over centuries and are not apparent without data sets of a century or more. Droughts, so apparent in their ecological effect on the terrestrial landscape, are also important to the aquatic environment, but their effects are usually much less visible. Major regional droughts, for example, are often correlated with increased upwelling of deeper, nutrient-rich ocean waters that stimulate increases in fish production. Even in the absence of all human activity, climate shifts cause the size of salmon runs to vary year-to-year and decade-to-decade.

EVOLUTION OF FISHERIES MANAGEMENT CONCEPTS

The history of fisheries management reflects the conventional wisdom of the day. During the past two centuries, the level of ecological understanding has greatly increased. Prior to the 1800s, most people presumed that biological resources from inland and marine waters were inexhaustible, given the level of harvest possible by the modest number of people who fished and the limited effectiveness of their fishing gear.

By the mid-1800s, the idea of unlimited natural riches from inland waters and the ocean was no longer credible. One popular approach to overcoming nature's constraints on sustainable harvest was to apply *animal husbandry* concepts, including the idea that "seeding nature" with fish produced in captivity would permit much greater levels of fishing. Conventional wisdom held that *aquaculture* could produce a nearly unlimited supply of fish of superior quality and according to a predictable schedule, just as farmers had long achieved for domestic livestock. In captivity, fish could be fed a high quality diet, protected from predation, and the quality of the product improved by selective breeding. If aquaculture performed as hoped, fisheries managers would no longer have to depend on the vagaries and limitations of nature for fisheries products.

For many species in many situations, aquaculture has worked well. Culture techniques greatly improved in the late 1800s. Selective breeding created animals better adapted to life in captivity. However, the expectation that

aquaculture would be a solution to nature's limitations was not fully realized.

By the early 1900s, the limitations of aquaculture as a tool to supplement or replace wild fish were being recognized and *harvest regulation* was considered the more effective way to ensure sustained harvests. However, because the scientific underpinnings for many harvest regulations were poor and public pressure to continue heavy fishing was great, regulations were often modest, poorly enforced, and produced disappointing results in limiting harvests to sustainable levels.

As the recognition increased the need for regulations to control overfishing, an appreciation emerged that *habitat* was a limiting factor in fish yields, especially in inland and coastal waters. Thus, efforts to improve stream, lake, and estuarine habitats became more common throughout North America and continue today.

By the mid-1900s, *scientific* fisheries management was the dominant paradigm. The idea underlying this approach was that every fish population had the potential to produce a harvestable surplus and the largest surplus that could be harvested annually from that population (*maximum sustainable yield*) could be estimated by rigorous scientific analysis (stock assessment). The job of the fisheries manager was to control fishing pressure, using various regulations, at a level such that sustainable catch levels could be achieved in perpetuity. However, fishing pressure, as always, was very difficult to control; many fisheries ended up being overharvested, and yields eventually declined.

In the 1970s, the concept of *optimum sustainable yield* became popular, primarily in response to concerns that efforts to maximize the catch in recreational fisheries management overlooked, or at least undervalued, many important benefits that fishermen and society received. Less commonly, it was also used in managing commercial fisheries. Management goals using this approach tended to weigh more heavily the *quality* of the fishing *experience* or the *socioeconomic* aspects of fishing and to place less emphasis on the actual catch. Maximum benefits to society were usually achieved at catch levels *below* the maximum sustainable yield.

By the late 1900s, the trend in recreational fisheries management was toward *species* and *habitat* protection, especially in inland and coastal waters. The widespread recognition that some aquatic species were at risk of extinction led to public pressure to reverse such trends. The main causes of the decline of fish species was habitat alteration and introduction of nonnative fish species. Only rarely was overfishing the primary cause of precipitous declines in fish abundance. In fact, most endangered fish species have never been fished. "Endangered species" and "species at risk" legislation directed government agencies and fisheries managers to emphasize protecting species above catch. Under such legislation, fishing may be permitted only if it does not jeopardize legally protected species.

A recent trend in recreational and commercial fisheries management has been the emergence of the *stakeholder approach*. A stakeholder is any citizen or group potentially affected by, or having a vested interest in, an issue,

program, action, or decision. The idea behind involving stakeholders in fisheries management is that society has a wide range of conflicting views on what fisheries management goals should be; therefore, it is desirable to include input from the full range of stakeholders in the process of defining goals and selecting management measures. Fisheries management plans developed with stakeholder involvement, it is assumed, therefore, have a higher likelihood of garnering widespread support.

Comanagement represents a further development of the stakeholder approach where some of the authority to manage the fishery is vested in the fishermen themselves or in organizations such as Indian or tribal governments. There are many current and proposed variants of comanagement, but they all transfer a degree of regulatory authority from government fisheries agencies to fishermen, associations, or other organizations. In some fisheries, a market is created for individual, transferable, fishing rights. In such situations, the right to fish may be purchased in an open market.

A widely accepted development in fisheries management has been the *precautionary principle*. The basic concept is that decision-makers (e.g., fisheries managers) ought to err on the side of caution in managing natural resources. Often the scientific basis for fisheries management decisions contains considerable uncertainty. Given the highly unpredictable future environmental and social conditions, it is wise for managers to use caution in managing fisheries.

LAKE FISHERIES MANAGEMENT

Lakes (including reservoirs formed by man-made dams) vary from large (Great Lakes, Caspian Sea) to small (farm ponds, minuscule alpine pools).

Maintaining at least reasonable good water and habitat quality is absolutely essential to nourishing healthy fish populations. Pollution control and abatement, although typically outside the direct purview of fisheries managers, are essential if management goals are to be achieved.

For large lakes, fisheries management involves primarily assessing, then selecting, fishing or harvest levels that are sustainable. Other management techniques include *intentional* introduction of nonnative species (e.g., Pacific salmon to the Great Lakes), or control of *unintentional* and undesirable introductions (e.g., sea lampreys in the Great Lakes).

There are more management options for small lakes and reservoirs. Small lakes may be manipulated by altering water levels or habitats (e.g., improving spawning areas, adding brush piles to provide hiding places for fish) or altering water quality (e.g., fertilizing low nutrient lakes, reducing the flow of nutrients into lakes that have excessive nutrients) to increase the productivity of desirable species. In some circumstances, improving access to the lake may promote greater use by fishermen. In extreme cases, small lakes may be *chemically rehabilitated* (e.g., all fish are removed by complete poisoning and a desirable mix of fish species reintroduced). Regulation of lake fishing usually involves limits on the *catch* (species, number, and size that may be kept), *time* (season and

hours when fishing is permissible), and *gear* (type of fishing equipment and bait that may be used).

Reservoirs often present additional challenges to fisheries managers because they are built for other primary purposes (e.g., flood control, electricity generation, irrigation, water storage, transportation), and these uses often conflict with secondary purposes, such as achieving fisheries benefits for society. Water level fluctuations in many reservoirs, both daily and seasonally, which often result from electricity generation, flood control, and irrigation practices, have profound consequences for reservoir fish populations. In practice, fisheries managers must work collaboratively with many other groups to achieve a mix of societal benefits. Only a few of the benefits from most reservoirs are associated with fish, fishing, and environmental quality.

Management of many small lakes tends to be the responsibility of a single governmental or nongovernmental entity, greatly simplifying fisheries management. Large lakes, on the other hand, tend to have more complex managements, involving multiple agencies. Such inter-jurisdictional decision-making greatly compounds fisheries management problems. For example, because five nations surround the Caspian Sea, managing the sturgeon fishery (which produces highly valued caviar) sustainably has been difficult due to the lack of a single, enforceable management plan. As a result, Caspian Sea sturgeon populations have dropped 90% during the past several decades.

RIVERINE FISHERIES MANAGEMENT

Riverine systems describe a continuum of aquatic systems ranging from small creeks a few miles long to rivers the size of the Amazon, Mississippi, McKenzie, Yukon, and Columbia. One important characteristic is that many riverine systems pass through, or form the boundaries between, different political jurisdictions. Multiple political and management jurisdictions often result in unsatisfactory management of riverine fisheries when agencies fail to cooperate toward achieving common societal goals.

Rivers, especially larger ones, commonly undergo extensive habitat alteration resulting from human activities such as building dams, dikes, bridges, shipping channels, and waste treatment plants. Also, because of dikes and other structures, it is common to lose the connections between rivers and their productive flood plain channels and backwaters, habitats that often provide essential spawning and nursery areas for fish. In many cases, fisheries managers must work with highly altered ecosystems that are no longer suited for the fish species most valued by the public.

Harvest regulations are important in river fisheries, but habitat protection and improvement are especially important. Many rivers, both large and small, are severely polluted or altered by domestic, farm, and industrial waste, agricultural and urban runoff; water withdrawal for domestic, agricultural, and industrial use; siltation; and riparian (streamside) alterations. Poor water quality caused by pollution and lack of habitat diversity are

often the limiting factors in developing successful fisheries management programs in rivers. The effects of pollution may be very subtle and indirect. Certain pollutants, for example, may make fish more vulnerable to predation by slightly reducing their ability to sense the presence of a predator.

Unlike lakes, streams and rivers are flowing systems that have a notable self-cleansing ability. Rapid turnover of water can lead to faster recovery of water quality than that in lakes or wetlands, where nutrients and pollutants may remain trapped in sediments for many years. However, the lower, coastal sections of rivers are often slow-moving and are often long-term repositories for contaminants.

Habitat alterations that adversely affect fish are common in riverine systems. The Colorado River of the southwestern United States, for example, is subject to numerous flow diversions and is reduced to a mere trickle by the time it reaches the Gulf of California. The Columbia River and its tributaries, arguably the most regulated river system in the world, contain 250 “large” dams and several thousand “small” ones. Such highly altered habitats no longer favor native fish species, many of which are migratory. Under these altered habitat conditions, certain nonnative fish species may prosper, and many native species are reduced in number or even extirpated.

Small riverine systems (streams, brooks, and creeks) are especially vulnerable to overfishing, habitat destruction, and the effects of land-use practices such as farming and urbanization. In extreme cases, management may include prohibition of fishing or, at least, highly restrictive fishing regulations. At the other extreme, some large riverine systems, especially in their lower reaches, may be managed similarly to large lakes and coastal fisheries. Commercial fishing may be the dominant type of fishing in many large rivers, especially in tropical areas.

COASTAL FISHERIES MANAGEMENT

The marine environment immediately adjacent to land supports *coastal fisheries* and includes nearshore marine, estuarine, and intertidal ecosystems. An *estuary* is a coastal waterbody that has a free connection to the ocean, and alternately with the tides, exhibits characteristics of both fresh- and saltwater environments. Because marine biota move into coastal rivers, the lower, intertidal reaches of rivers are often included as coastal fisheries. Intertidal environments are particularly important as nursery areas for the juveniles of many valuable saltwater fish species.

Coastal fisheries present an array of challenges to the fisheries manager. Human population density tends to be higher along the coasts, and this means that aquatic coastal habitat is likely to be substantially altered (e.g., sea walls, dredging, draining, and buildings) or polluted (e.g., from municipal and industrial waste, ship discharge, and runoff). One of the greatest challenges in managing coastal fisheries is the loss of coastal wetlands. These wetlands provide habitat for many adult fish and shellfish and are also essential breeding and rearing areas.

Coastal fisheries are also often heavily harvested, and there tend to be serious conflicts among user groups.

Surf fishermen, crabbers, shrimp trawlers, oil extractors, shippers, boaters, swimmers, and sightseers all use the coastal environment in ways that often conflict with each other. Use can be intensive. There are more than 9 million salt water recreational anglers in the United States.

A major change in coastal fisheries management began in the 1970s as some nations extended their offshore management jurisdiction in an attempt to control fishing by foreign nations. In 1982, most nations adopted the United Nations *Law of the Sea Convention*, which recognized the 200-mile line separating the high seas from waters in the *exclusive economic zone* of the adjacent nation. If desired, individual nations could extend their exclusive economic zone to 200 miles under international law. The United States did so in 1983. In spite of the Law of the Sea Convention, overall fishing pressure has generally remained heavy, especially in exclusive economic zones, because domestic fleets soon replaced foreign fleets.

OPEN OCEAN OR HIGH SEAS FISHERIES MANAGEMENT

Open ocean fisheries are those that operate away from the coasts and often outside of any nation's territorial waters. The two general categories of fish that are targeted in such fisheries are the *pelagic*, or open-water-dwelling, fish and the *demersal*, or bottom-dwelling, fish. Pelagic fish species tend to feed and travel near the ocean surface. Demersal fish species tend to live on the continental shelves closer to shore. Tuna and swordfish are examples of commercially important pelagic species; cod, hake, flounder, and toothfish are important demersal species.

The inadvertent capture of nontarget species (called *bycatch*) is a serious management challenge in many fisheries but especially for open ocean and coastal fisheries. Perhaps a third (sometimes much more) of the catch is discarded by fishermen as not marketable. Bycatch may be the young of valued sport or commercial fish species or important food sources for sport or commercial fish species. Shrimp trawlers, for example, catch and discard large quantities of small fish while pursuing the much more valuable shrimp. Various types of fishing gear used by commercial fishermen can injure, and often kill, protected animals such as seabirds, marine mammals, and sea turtles. For example, the indirect catch of dolphins in tuna fishing has led to consumer boycotts and a demand for “dolphin-safe” tuna products. Likewise, devices that effectively exclude turtles from capture have been incorporated into the trawls used by commercial shrimp fishermen.

Habitat alteration caused by certain fishing gear is also a concern in some locations. Trawling (trawls are large, heavy nets dragged by fishing boats) may alter the physical and biological characteristics of the seabed, in particular sea grass beds and coral reefs, in ways detrimental to the well-being of target fish populations. Thus, fisheries managers may have to balance how to minimize the habitat alteration of sea floors caused by trawls while still permitting capture of the target species.

Serial depletion of fish populations is another challenge to managers of open ocean fisheries. Typically, this means that fishermen move to new fishing grounds as those closer

to home are depleted. A related type of serial depletion is that caused by the development and use of improved fishing gear that allows fishermen to exploit new fishing grounds as the old ones are depleted. A recent trend has been for open ocean fishermen to move into fishing more deep water environments as more accessible nearshore stocks decline.

Open ocean fisheries management is currently in a state of flux; intense international efforts are being exerted to make them economically and ecologically viable within a framework of producing sustainable, but profitable catches. For example, one challenge for fisheries managers is the heavy subsidies many nations provide to commercial fishermen which creates *excess fishing capacity*. In many cases, fishing would not otherwise be profitable, but subsidies in the form of tax incentives or cash payments make it cost-effective for fishermen to continue fishing. Thus, in many cases, the laws of supply and demand that would tend to prevent overfishing do not come into play.

DIADROMOUS FISHERIES MANAGEMENT

Diadromous fish are those characterized by a life cycle of either spawning in freshwater environments and spending their adulthood in marine environments (*anadromous* species) or spawning in marine environments and spending adulthood in freshwater environments (*catadromous* species). Diadromous fisheries represent unique challenges to fisheries managers because target species often cross multiple jurisdictions. Unless management efforts are well coordinated, the combined effect of decisions by different jurisdictions can result in serious depletion of these resources.

Anadromous fish (e.g., salmon, American shad, striped bass, smelt, and sturgeon) are important species commercially and recreationally. Many rivers no longer support major spawning runs of anadromous fish.

Until the 1800s, large runs of Atlantic salmon were found in many coastal rivers of both western Europe and eastern North America. By the middle to late 1800s, salmon runs in the eastern United States and western Europe had been drastically reduced by the effects of overfishing, dams, and pollution. Overall, runs continue to be much reduced on both sides of the Atlantic. The largest remaining runs, although small by historic standards, occur in eastern Canada, Iceland, Ireland, Scotland, and the northern rivers of Norway. Aquaculture has largely replaced harvested wild fish as the source of Atlantic salmon for the retail market.

The seven species of Pacific salmon found on both sides of the North Pacific also have, overall, declined significantly from historic levels, but not as dramatically as Atlantic salmon. Hatchery production has been used to maintain some runs in the southern region of the range (e.g., Japan, Korea, California, Oregon, and Washington). In California, Oregon, Washington, Idaho, and southern British Columbia, runs have been depleted by past overfishing, dam construction, water withdrawal for irrigation, competition with hatchery-produced salmon, competition with various nonindigenous fish species, predation by marine mammals and birds, and climatic and

oceanic shifts. Runs in the northern half of the range (e.g., Russian Far East, Alaska, Yukon, and northern British Columbia) are in much better condition. The northern runs have been abundant for the past several decades, but are likely to decline somewhat for several decades because ocean conditions in the North Pacific tend to shift on such a several-decade time cycle.

Striped bass, native to the East and Gulf coasts of North America, have been introduced to the Pacific Coast and are now found from Baja California to British Columbia. Overall, the species in its original range is less abundant than it was historically, but, nevertheless, catches are still substantial. Some runs on the west coast of North America do very well. The causes of the decline in its original range are similar to those that precipitated the drastic declines in salmon runs (i.e., dams, water diversions, pollution problems, and overfishing).

American shad are found from the Gulf of Mexico up the Atlantic coast and as far north as New Brunswick. Generally, in their native range, shad runs are much reduced from historic times due to dams, pollution, and overfishing. American shad have been introduced to the west coast of North America and have done well in some rivers, especially the Columbia and Sacramento—San Joaquin.

Several species of anadromous sturgeon are also of particular concern. Some are highly prized for their roe (eggs often sold as caviar) and must be carefully managed to avoid overfishing. Other sturgeon species are at risk of extinction, and drastic national and international measures may be required to protect these species.

Many other species have anadromous forms (e.g., smelt, alewife, blueback herring, and cutthroat, rainbow, brown, and brook trout) that support substantial fisheries in certain locations. In other locations, they are important in fisheries management because they are at risk of local extinction. Like salmon, all of these anadromous fishes are extremely vulnerable to dams and other impediments to migration; they are also sensitive to water diversions and pollutants.

A few species, such as American eels, are *catadromous*—they spawn in the ocean, but live their adult lives in freshwater. Eels are important commercial species in certain regions. They occur in rivers, lakes, estuaries, coastal areas, and open ocean. Their distribution ranges from the southern tip of Greenland, along the coast of North America, the Great Lakes, the Gulf of Mexico, the Caribbean, and as far south as northern South America. Most of the catch is exported to Europe and Asia.

AQUACULTURE

Aquaculture continues to expand its importance in both commercial and recreational fisheries management. The growth of aquaculture has been especially rapid during the past decade.

Food production is the most common objective in aquaculture. Commonly raised species are carp, Atlantic salmon, rainbow trout, catfish, tilapia, shrimp, oysters, mussels, and seaweed. Currently, one-third of the total world food fish supply is obtained from aquaculture, and

this portion is increasing. The amount of farmed fish produced worldwide has more than doubled since 1989.

Producing fish in captivity for subsequent stocking to enhance, maintain, or initiate fishing is also important, especially in North America. Trout are commonly raised in the United States and Canada to support fisheries subject to intensive recreational fishing, the so-called “put-and-take” fisheries. On the Pacific coasts of North America and Asia, commercial enterprises operate hatcheries to maintain runs of salmon artificially to meet market demand, a practice called *ocean ranching*. After being hatched and reared from eggs, juvenile salmon are released to migrate to the ocean, spend several years growing to adult size, then return to the hatchery of origin to spawn. At the hatchery, they are captured, and some are then spawned artificially to obtain eggs for the next generation. The rest are processed for market.

A fairly recent development in fisheries management, *conservation aquaculture*, uses aquacultural techniques to produce fish threatened by extinction. These “captive breeding” efforts may be the only hope of preserving or recovering certain fish species or populations when natural reproduction is compromised.

SPECIES AND HABITAT PRESERVATION

By the 1990s, management objectives for many freshwater fisheries in North America had shifted from optimizing commodity output to protecting habitat or preserving imperiled species. Concerns about loss of *biological diversity* and *biological heritage* often eclipsed concerns for sustaining commercial or recreational catches. This was especially true for the Pacific coast salmon fisheries and fisheries in more remote, pristine areas such as national parks and wilderness areas.

Overall, alteration of aquatic habitat (e.g., dam construction, flood control structures, dredging to facilitate water transportation, filling to create useable land, sediment and pollution runoff, and acid rain) is one important cause of the tenuous status of many fish species. Thus, protecting fish *habitat* has become a prime focus of many fisheries management agencies.

Nonindigenous fish species (those not native to the area) include both *exotic* species (those from a foreign land), and *nonnative* species (those that have expanded beyond their native range), and they often adversely affect valued native fish populations. Nonindigenous species often compete with or prey upon commercially or recreationally important fish species. They may also hybridize with closely related native species and cause a distortion in the gene pool. Nonindigenous species have contributed to the decline of approximately two-thirds of the threatened or endangered fish in the United States. Many nonindigenous fish introductions have been the result of intentionally releasing bait fish after a day's fishing, unintended releases from international shipping activities, and releases and escapes from the aquaculture and aquarium trades.

Not all nonindigenous fish species are perceived as management problems. Many highly valued and heavily harvested fish species were intentionally introduced by

fisheries managers and continue to enjoy widespread public support. Among the most widely introduced fish species in North America are Pacific salmon; rainbow, brown, and brook trout; striped bass; walleye; small- and largemouth bass; and bluegill. Recent trends in fisheries management have been away from introducing fish outside their native range.

Legislation to protect species from extinction also has affected the management priorities of fisheries agencies. For example, the U.S. Marine Mammal Protection Act (1972) and the U.S. Endangered Species Act (1973) are now the legal drivers for management of some fisheries (e.g., over much of the range of Pacific salmon in the United States, the primary management goal is to prevent extinction, not to increase catch).

Internationally, the *Convention on Biological Diversity* (1992) imposes legal obligations on all signatory countries to conserve their biodiversity, manage their fisheries resources in a sustainable manner, and promote fair and equitable distribution of the benefits of each nation's genetic and biological resources.

ECOSYSTEM MANAGEMENT

Beginning in the 1980s, a widespread view emerged that managing fisheries should be broadened in scope to include the entire ecosystem; hence, the rise of *ecosystem management*. A precise, universally accepted definition of ecosystem management has yet to emerge, but it is generally seen as the application of ecological, economic, and social information, options, and constraints to achieve desired social benefits within a defined geographic area and for a specified period.

Part of the appeal of ecosystem management is that it may better balance the suite of benefits (e.g., food fish, recreational fish, preserving endangered species, and preserving ecosystems) that society values. To date, ecosystem management has been most commonly implemented in public forests in North America. Efforts are now under way to apply the same concept to large lakes and open ocean ecosystems.

Adaptive management, the process of improving management effectiveness by learning from the results of carefully designed decisions or experiments, is often included in ecosystem management frameworks. The philosophy underlying adaptive management is the recognition that the ecological consequences of many fisheries management decisions are too uncertain to predict with confidence. Therefore, management decisions ought to be tentative and used to learn how the ecosystem responds. The information derived from such decisions allows the manager to adapt future decisions to reflect what has been learned from past decisions.

Another trend in fisheries management, now commonly practiced in Europe, is using the river basin as the management unit. River basins, or watersheds, have long been used in water management and pollution abatement but have only recently been adopted in fisheries management. River basins tend to be the preferred geographic level of fisheries management that international organizations use.

FUTURE OF FISHERIES MANAGEMENT

In the future, fisheries management will continue to reflect the overall values and preferences of the society within which it operates. Fisheries managers will strive to produce sustainable benefits from renewable biological resources, but society's needs will continue to evolve, resulting in different, and often conflicting, management goals. Efforts to maintain or increase the catch are likely to be tempered by society's growing interest in protecting the environment and preserving imperiled species.

Fish and fishing will remain important factors in the daily lives of many people but especially so in those areas where animal protein from agriculture is in relatively short supply. Overall, harvest pressure on most aquatic environments will continue to increase in concert with rising demand for animal protein for use as human and animal food and for recreation opportunities.

Harvest restrictions in many fisheries are likely to become more constraining as fisheries managers attempt to maintain sustainable yields and avoid fishery collapses. International trade in fisheries products is likely to be scrutinized to a greater degree in response to perceived environmental damage caused by excessive or inappropriate fishing. To counter past overfishing in the ocean, for example, there is likely to be increasing public pressure to create legally protected areas where fishing is forbidden (e.g., fish parks, marine reserves, and marine sanctuaries).

Protecting fish habitat will continue as a primary management goal in the future. There will be a continuing emphasis on protecting the environment in general and water quality in particular. The emphasis in enhancing water quality is likely to shift from controlling specific sources of pollution to reducing pollution from large-scale runoff. Fisheries managers will increasingly be focusing on water quality enhancement and pollution abatement. Introduction of nonindigenous species and genetically altered fish will be of increasing concern to fisheries managers.

In the future, aquaculture will likely provide a larger percentage of the total worldwide production of fisheries products. Domestication, genetic selection, genetic engineering, and other technological developments are likely to be employed to enhance aquacultural production.

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FACTORS AFFECTING FISH GROWTH AND PRODUCTION

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The health and subsequent growth of fish are directly related to the quality of water in which the fish are raised. In general, factors affecting fish growth and production in freshwater aquatic systems can be classified as physical, chemical/biochemical, or a combination thereof. The physical properties of water that are important to fish production and growth include temperature and the concentrations of suspended and settleable solids; important chemical parameters include pH, alkalinity, hardness, and metals (e.g., iron, aluminum, calcium, etc.) The solubility of oxygen and ammonia gases, which vary as a function of other parameters such as temperature, are also key parameters in maintaining fish health. Recommended water quality criteria for salmonids are

Table 1. Summary of General Recommended Water Quality Criteria for Salmonids^a

Analytic Parameter	Water Quality Criteria
pH	6.5–8.0 standard units
Total hardness and alkalinity	10–400 mg/L as CaCO ₃
Calcium	<160 mg/L
Magnesium	<15 mg/L
Iron	<0.1 mg/L
Aluminum	<0.01 mg/L
Manganese	<1.0 mg/L

^aReference 1.

presented as typical chemical requirements of freshwater fish in Table 1 (1). However, note that specific water quality requirements are highly species dependent and further references should be consulted.

TEMPERATURE AND DISSOLVED OXYGEN (DO)

Water temperature is one of the most important physical factors affecting fish growth and production. Fish are cold-blooded animals which assume approximately the same temperature as their surroundings. Typically, fish are classified broadly as cold, cool, or warm water, depending on their tolerance for particular temperature ranges. Within each temperature classification, fish survival is bounded by an upper and lower temperature, between which an optimum temperature for growth exists. When temperatures vary outside the optimum range, decreased tolerance to changes in water quality constituents (particularly metabolites such as ammonia) and a decrease in immunological response can occur. Consequently, decreased growth/productivity and in some cases, mortalities, may result, depending on the magnitude of the deviation from the optimum temperature. For example, the survivable temperature range for rainbow trout (*Oncorhynchus mykiss*) is ~1 to 26°C. However, depending on the reference source, the “optimum” temperature range for growth is 13°C to 21°C (1).

Representative optimum temperature ranges for common freshwater fish species are presented in Table 2. The survivable and optimum temperature ranges vary among species, so readers are referred to the comprehensive enumerations of temperature limitations for many freshwater fish species presented by Colt and Tomasso (2), Boyd and Tucker (3), Tomasso (4), Barton (5), Meade (1), and Andrews and Stickney (6).

Water temperature can also affect the solubility of gases which are important to fish health (e.g., dissolved oxygen and ammonia). In general, the solubility of gases decreases when water temperature increases because the chemical species have a greater affinity for the gas phase than the aqueous phase (7). For example, the dissolved oxygen concentration over a range of temperatures is presented in Table 3 (8).

Dissolved oxygen concentrations near saturation (≥95% saturation) are recommended for fish health and productivity (1). Further, the addition of oxygen to fish culture

Table 2. Representative Optimum Temperature Ranges for Common Freshwater Fish Species^a

Name	Optimum Temperature Range, °C
Coho salmon (<i>Oncorhynchus kisutch</i>)	11–17
Rainbow trout (<i>Oncorhynchus mykiss</i>)	13–21
Smallmouth bass (<i>Micropterus dolomieu</i>)	18–24
Walleye (<i>Stizostedion vitreum vitreum</i>)	20–23
Channel catfish (<i>Ictalurus punctatus</i>)	25–30
Talapia (<i>T. mossambica</i>)	28–30
Bluegill (<i>Leptomichthys macrochirus</i>)	29–32

^aReference 1.**Table 3. Dissolved Oxygen Concentration Over a Range of Temperatures^{a,b}**

Water Temperature, °C	Dissolved Oxygen Concentration, mg/L
0	14.62
5	12.80
10	11.33
15	10.15
20	9.17
25	8.38
30	7.63

^aReference 8.^bUnder standard conditions in fresh water (Cl⁻ concentration = 0 mg/L).

waters via aeration systems can aid in reducing the toxicity of un-ionized ammonia by driving the volatile gas off through mixing/aeration (9). Due to the toxicity of ammonia to fish, nitrogen species are presented and discussed later in a separate section.

SOLIDS

Solids in water can be in either suspended or settleable form; each can have both direct and secondary effects on the ability of water to support fish. High suspended solids concentrations, measured in the field as turbidity, can irritate fish gills and thus, lead to respiratory deficiencies in fish. Further, suspended solids can decrease water clarity and thus photosynthesis, which can lead to adverse effects on aquatic vegetation and consequently, on herbivorous fish species. High sedimentation in natural waterways can smother fish eggs. Similarly, sedimentation can degrade benthic macroinvertebrate habitats and thus have substantial adverse impacts on insectivorous fish species (10).

Organic matter is also often associated with the solid phase; thus, high suspended and/or settleable solids concentrations can indicate sources of increased oxygen demand in the water which may then lead to oxygen deficiencies for fish. Further, a direct linkage between organic solids loading and bacterial gill disease, as well as amoebae gill infestation, has been reported (11); solid-phase organic matter serves as a host site for opportunistic pathogens.

To control the effects of solids adequately on fish growth and productivity, it is recommended that total solids concentration be maintained at ≤ 80 mg/L. A comprehensive overview of solids removal technologies is presented in Metcalf and Eddy (12) and Wedemeyer (13). Additional representative works related to specific solids management technologies in fish culture systems include Viadero and Noblet (14), Kristiansen and Cripps (15), Summerfelt (16), and Chen et al. (17).

pH

In general, a pH of 6–9 is needed to support aquatic flora and fauna (7), though specific requirements are species-dependent. For instance, rainbow trout (*Oncorhynchus mykiss*) are relatively intolerant to acidic conditions, whereas green sunfish (*Lepomis cyanellus*) and *Rhinichthys atratulus* (black nose dace) are often found in waters recovering from impairment related to acid drainage (10). Other concerns related to fish growth as a function of pH include increases in toxicity of aluminum and ammonia at elevated pH (18,19).

ALKALINITY AND HARDNESS

Carbonate alkalinity is a measure of water's ability to neutralize strong acids due to the presence of carbonate species and is thus an important factor in buffering waters against abrupt changes in pH (7). In natural waters, carbonate alkalinity is derived mainly from the dissociation of $\text{CaCO}_3(\text{s})$ and is consequently, directly related to water hardness. Further, elevated water hardness decreases the toxicity of metals such as copper and zinc through the formation of soluble metal–inorganic ligand complexes which are typically not as readily bioavailable to fish as free metal ions (7,18,20–23). A definitive recommended limit for total hardness in fish culture waters is not clear (22), though Heath (24) and Meade (1) noted positive effects on fish health in waters with hardness in excess of ~ 400 mg/L as CaCO_3 . In contrast, low alkalinities generally only contributed to fish stress when concentrations of CO_2 , a weak acid often found in groundwater, were elevated.

DISSOLVED METALS

The metals of greatest concern for fish growth and production include copper, zinc, tin, cadmium, mercury, chromium, lead, nickel, arsenic, and aluminum (24). The USEPA (25) has listed several of these elements as priority water quality pollutants for human consumption of fish, including mercury, cadmium, and selenium. Heinen (18) synthesized the USEPA “priority elements” along with other metals of concern for effects on fish growth and production, as presented in Table 4.

In addition to the priority pollutants listed in Table 4, other metals, including iron and aluminum, can have adverse effects on fish productivity and growth. For

Table 4. Prioritized List of USEPA “Priority Elements” and other Metals of Concern for Effects on Fish Growth and Production^a

Analytic Parameter	Water Quality Criteria, Listed in Order of Importance
High	Mercury, cadmium, selenium
Medium	Arsenic, lead
Low	Chromium, copper, zinc, antimony, nickel, silver, beryllium, thallium

^aReference 18.

instance, both ferrous and ferric iron (Fe^{2+} and Fe^{3+} , respectively) can be toxic to fish. Ferrous iron is reportedly more toxic to fish than ferric iron, though in well-aerated waters, Fe^{3+} is the dominant form of dissolved iron. Ferrous iron, however, may be detrimental to fish health if precipitated in large quantities as $\text{Fe}(\text{OH})_2(\text{s})$, as it may “clog” fish gills (18).

Aluminum is toxic to rainbow trout (*Oncorhynchus mykiss*) at lower concentrations than those of other metals because it can accumulate on the gills as aluminum hydroxide and cause respiratory impairment (18). In addition, aluminum is highly toxic to trout in both dissolved and suspended forms; however, the symptoms of poisoning are different, depending on the form/phase of aluminum. Dissolved aluminum tends to cause acute mortalities, and suspended aluminum results in more chronic effects such as respiratory difficulties indicated by gill hyperplasia (26). It has been recommended that concentrations of total aluminum (both dissolved and suspended) be maintained at less than 0.01 mg/L for normal growth and survival of trout. However, the effects of complexation between aluminum and inorganic ligands may abate the effect of high dissolved aluminum concentrations by rendering a fraction of aluminum less bioavailable than free Al^{3+} ions (27).

Criteria for calcium and magnesium concentrations in trout waters are based on limited data (18); common recommendations include calcium concentrations ≤ 160 mg/L (22) and magnesium concentrations ≤ 15 mg/L, though an alternative limit of 28 mg/L Mg has also been suggested by (28).

NITROGEN SPECIES

Total ammonia nitrogen (TAN) is comprised of two species: (1) un-ionized ammonia (NH_3), which is highly toxic to fish and (2) ionized ammonia (NH_4^+), which is not believed to be toxic to fish (29). The widely accepted TAN limit for “no effect” is 0.0125 mg/L $\text{NH}_3\text{-N}$ (29–31).

Long-term exposure to subacute un-ionized ammonia concentrations can decrease productivity (29,32) and change gill structure which may result in death from oxygen deficiency (11,33,34). Reduced growth of rainbow trout was observed at a concentration of 0.0166 mg $\text{NH}_3\text{-N/L}$. However, fish may be able to maintain sufficient oxygen uptake by changing cardiovascular and respiratory behaviors, though activity would decrease (35,36).

Water chemistry can also impact ammonia toxicity. For example, it has been widely reported that increased salinity and divalent cation concentrations may reduce the toxicity of $\text{NH}_3\text{-N}$ by altering the osmoregulatory systems of fish and decreasing membrane permeability (29,37,38). Soderberg and Meade (29) studied acute and chronic toxicity to determine the effect of calcium and sodium on the toxicity of un-ionized ammonia to Atlantic salmon (*Salmo salar*) and lake trout (*Salvelinus namaycush*) and found that ionic strength was important in determining the effect of $\text{NH}_3\text{-N}$ exposure on different species and sizes of fish. Temperature and pH were also significant water quality parameters because they affect the solubility and speciation of ammonia in solution. Methodologies to account for the effects of pH, temperature, and ionic strength are presented in Soderberg (9) and Soderberg and Meade (39).

Nitrite (NO_3^-) is produced by *Nitrosomonas* sp. through the biochemically mediated oxidation of ammonia. Nitrate concentrations in excess of 0.55 mg/L can result in methemoglobinemia, a condition in which the iron in blood cannot transport oxygen. As a consequence, oxygen uptake by fish becomes limited.

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WATER NEEDS FOR FRESHWATER FISHERIES MANAGEMENT

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THE SCOPE OF FISHERIES MANAGEMENT

“Fishery management” is variously defined, but common to all definitions is manipulating fishery resources to produce sustainable benefits for people. Fishery management relies on principles of aquatic ecology but focuses on the use of fishery resources.

A “fishery” has three components: the fish, the habitat, and the users (people). Effective fishery management embraces all three components. It is important to recognize that “the users” often are people who catch fish for sale or recreation, but the users are not necessarily consumptive. The thrust of this article, the dependency of fishery management on water, necessitates a focus on the habitat component of fisheries. However, no component is completely independent of the others, and the linkage between people and habitat is especially influential on fishery management.

WATER QUALITY

Good water quality is essential to fish survival, growth, and reproduction. Most fish require dissolved oxygen concentrations above 5 mg/L (parts per million), and 3 mg/L is considered stressful for many warm-water fish. Brief periods of hypoxia can be lethal. Fish survive over a pH range of 4.5–10, and acidic inputs from mine drainage

or precipitation can be lethal. Freshwater fish vary in their tolerance of salinity; some species can survive in seawater, but others can tolerate less than 10% seawater. Many heavy metals and pesticides and their derivatives are toxic to fish, but some chemicals, such as mercury and PCBs, are accumulated by fish to levels toxic to other animals, particularly mammals and birds. Ammonia and nitrite can be toxic at less than 2 mg/L. Some evidence is now available that hormones in discharge water can disrupt endocrine function and interfere with fish reproduction. Tolerances of fish to water quality variables may vary among life stages; for example, eggs or larvae may tolerate much lower salinity or concentrations of a toxic chemical than adults.

Point-source pollution, although not abated, is relatively well controlled in most developed countries compared with nonpoint pollution—in essence, runoff from the watershed. Inputs of otherwise nontoxic and biologically essential chemicals, such as nitrates and phosphates, can have beneficial or benign effects on aquatic systems. However, excessive levels of these chemicals that result from agriculture, confined animal rearing facilities, and domestic and municipal development have dramatic effects on receiving waters and cause undesirable shifts in aquatic ecosystem structure and function. In this regard, it is important to recognize that many chemical inputs to aquatic systems are not directly injurious to fish but, by altering the composition and abundance of other biota, indirectly influence the health and survival of fish populations. A common example is oxygen depletion that results from planktonic algae blooms or excessive growth of aquatic plants (macrophytes). Although the plants usually produce the oxygen needed to sustain fish and other aquatic life, dense macrophytes or algae blooms fueled by high phosphorus concentrations can consume oxygen at night and during reduced-light conditions and cause fish kills. Oxygen depletion can also occur at the end of the growing season when excessively abundant plant material dies and decomposes.

WATER QUANTITY

It is obvious that fish need water, but there are aspects about water quantity that are far from obvious and are often overlooked by managers. In standing water systems, the amount of water impacts fish by affecting the types and amounts of habitats available and may alter thermal conditions. Declines in water level generally result in reduced areas of desirable aquatic habitat, and reduced water volume, in turn, impacts physicochemical conditions. Most freshwater fish spawn in relatively shallow water, and drawdowns at that time can cause spawning failures. The young of many fish find both shelter and food in shallow water, littoral zones (the nearshore area where rooted aquatic plants grow); dewatering these habitats reduces fish growth and survival and may result in year class failures. However, seasonal declines in water level can be desirable in shallow lakes where periodic drying of littoral zones is necessary to oxidize accumulated organic matter and consolidate soft, detritus-rich bottom sediments.

Rivers and streams are dynamic systems in terms of amount (discharge) and level (stage) of water, but the fish endemic to these systems are adapted to the annual cycle of fluctuations. Although peak and low flows and stages may vary from year to year, the annual cycle of rise and fall (the hydrograph) is rather predictable. Alteration of the annual hydrograph, such as may occur when a river is dammed for flood control or water supply, can interfere with spawning and recruitment by decoupling river elevation and thermal cycles, preventing access to essential habitats for floodplain-spawning fishes, or failing to provide necessary current velocity for fish that require flowing water for successful reproduction. River fluctuations are also important to adult stages of many fishes. Seasonal inundation of the floodplain provides access to rich food supplies for many fish, and receding waters flush nutrients, organic material, and biota, including small fish, into the river. As such, this "flood pulse" strongly influences fish production in river-floodplain ecosystems. High river stages also connect the river to floodplain lakes that were former river channels. Connection with the river can supply nutrients to the lake and allow in and out movements of fish. This connectivity, even if brief, unites the numerous different habitats in rivers necessary to sustain the diverse fish assemblage.

ALTERATION OF AQUATIC HABITATS

Water is essential to fish, but it is also essential to society and economic development. Natural river flows are altered for flood control, navigation, hydropower generation, water supply, and other anthropocentric purposes. Most of these purposes are accomplished by constructing dams to impound water and regulate flows. Dams are barriers to fish migrating to upstream spawning habitats and often curtail or even eliminate spawning. Further, the lake conditions created by dams eliminate habitats and flowing water essential to part or all of the life cycle of many native riverine fish. Reservoirs, by serving as sediment and nutrient traps and biological filters, can reduce the nutrients and sediments available to downstream systems. The resulting oligotrophication results in reduced fish production in downstream waters, a rather ironic reversal of escalating nutrient accumulation (eutrophication) common to many lakes and reservoirs.

Sedimentation is probably the greatest threat to river and stream fisheries resources. Although sedimentation is a natural process, excessive sediment input into waterways results from poor land management on the watershed. Sediment covers hard and rough bottoms essential for spawning and juvenile or adult stages of many fishes. The sediment smothers bottom-dwelling invertebrates; many are important food sources of fish; others are important to energy cycling in the stream. Sediment also provides an unstable substrate; thus, aquatic vegetation and bottom-dwelling animals that live in and on these soft substrates are often washed away during floods.

Natural thermal cycles are important to fish production, and "thermal enrichment" can be a form of pollution.

Thermal enrichment, usually resulting from discharge of cooling water from electric generating plants or industry, can benefit fish production by lengthening the growing season and may benefit users by lengthening the fishing season. Unfortunately, adverse impacts usually outweigh benefits. Elevated temperatures can stress kill fish, alter their metabolic requirements, alter predator-prey dynamics, cause out-of-season spawning, and result in replacement of cool- and cold-water fish with species tolerant of warmer water. Water temperatures lower than natural thermal conditions can also be problematic. Coldwater, hypolimnetic discharges from reservoirs can reduce aquatic production downstream, impact fish growth and recruitment, and alter the fish assemblage in favor of cold-tolerant species.

BIOTIC POLLUTION

Introduced aquatic species are an expanding and insidious form of pollution. Introduced species arrive in waterways by various means, including intentional but ill-advised introductions. The greatest source of aquatic nuisance species is intercontinental shipping. More than 160 nonnative species have been documented in the Great Lakes. Most, if not all, arrived in the ballast water of large ships. These alien species can alter water quality, change the pathways of energy and nutrient flow, alter habitats, prey on or compete with native species, hybridize with native species, and introduce diseases and parasites. Rarely are the effects beneficial; and most introduced species, once established, cannot be eliminated.

COORDINATED MANAGEMENT

Most public-access waters are multiple-use resources. As such, fishing is just one use of a waterbody, which is especially true for the more than 10 million acres of large impoundments in the United States. These systems were built to accomplish one or more purposes—hydropower, flood control, water supply, and navigation—invariably directed at achieving social or economic benefits. Fishing, although typically included in benefit analysis, was not a primary purpose for constructing large reservoirs. Fulfilling the primary purposes for which the reservoir was built dominates reservoir operation, often diminishing fishery values. In the last two decades, resource managers have become more sensitive to fishery needs.

SHIFTING TO MORE HOLISTIC MANAGEMENT

The study of inland waters (limnology) is only a little more than a century old, and modern fishery management has been developing for only half that long. As knowledge grows, perspectives and paradigms shift. Fishery managers now recognize the value of managing fish assemblages rather than single species. Furthermore, managers are realizing that lakes and rivers are not isolated microcosms but rather part of larger watersheds or ecosystems, and what happens on the watershed or in a land-water ecosystem affects what happens in the

water. Future management of fishery resources requires coordination and cooperation among diverse interests.

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AN OUTLINE OF THE HISTORY OF FISHPOND CULTURE IN SILESIA, THE WESTERN PART OF POLAND

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The need for pond construction in Silesia arose from the lack of large natural waterbodies and the long distance to the sea. The abundance of water, benign climate, fertile soils, the growing demand for fish in the developing towns; and the possibility of transporting fish along the rivers favored the development of fish farming.

The Silesian territory (Fig. 1) situated on the junction of Slavonic and German nations was affected by numerous wars, but at the same time, it was a center where various cultures existed (2). The political position of Silesia changed several times. From 990 to 1327–1329, it belonged to Poland. The foundation of the Wratislawia bishoprics by Bolesław the Brave, the Polish king of the Piast dynasty, in 1000 was an important moment in the history of fishpond culture because monasteries were the first to play an important role in fish culture in the Middle Ages. In the twelfth century, division among the sons of a Polish prince from the royal dynasty of Piast gave rise to many Silesian principalities.

In 1327–1329, the different political orientation of the Silesian princes caused it to affiliate with the Czech

Kingdom (Fig. 2). From the fifteenth century the southern part (Teschenensis and Oppoliensis-Ratiboriensis) was called Upper Silesia and the northern (Wratislawiensis), Lower Silesia. In 1526, Silesia became a part of the Habsburg Empire; in 1763 the entire Silesian territory was incorporated into Prussia; in 1918, it was partly affiliated with Poland, and in 1945, Silesia became part of Poland. Political, social, economic, and climatic variations distinctly affected the history of Silesian fishponds, which for ages played a manifold role in alleviating freshets and floods of rivers during wet periods and served as water resources during dry years.

This outline is based on secondary sources, chiefly the Polish literature including Czech, German, and Moravian citations. The historical data are derived from Encyclopedia PWN (1983–1987) (3).

THE FIRST PERIOD

The first development and breakdown of fishpond culture occurred between the twelfth and middle of the fifteenth century. Probably, Cistercians began to breed fish in ponds in the eleventh or twelfth century. Fishpond farming developed after the rivers and natural waterbodies had been wastefully exploited owing to the increasing population and strict observance of fasting. The highest progress in fish farming occurred in the fourteenth century and resulted from a rapid economic progress accompanying the growing mining industry and developing trade. Numerous fishponds as well as the introduction of the carp, probably from Czech and Moravia, allowed people to meet the religious demands. Devastating and depopulating wars, chiefly the incursion of the Tatars, obstructed fishpond development. The Hussite and Polish-Czech-Hungarian wars for the Czech throne ended the first period of development of fishpond culture in Silesia. Unfavorable climatic events were also responsible for this situation (4–6), although the outstanding development of fishpond culture might point to the fourteenth century as the warm climate phase.

THE SECOND PERIOD

In the middle of the fifteenth century, pond culture revived and developed till the end of the sixteenth century although some authors point to the sixteenth century as the beginning of the Little Ice Age (7). It resulted from a fairly peaceful time, warmed climate, popularization of carp culture and also due to privileges bestowed, mostly upon the oligarchy, by Ladislau of the Polish royal dynasty of Jagiellon, the king of Czech and Hungary.

Numerous ponds belonged to principalities and nobility, also to burghers, monasteries, and peasants. At the time of the highest development of pond fish culture, some places in the Silesian territory had the character of a lake district. Imagining the landscape at that time gives a map of a part of Upper Silesia elaborated in 1725 after a significant reduction in pond area in this territory (Fig. 3).

The fish from Upper Silesian ponds were transported by the River Vistula to Cracovia the capital of Poland

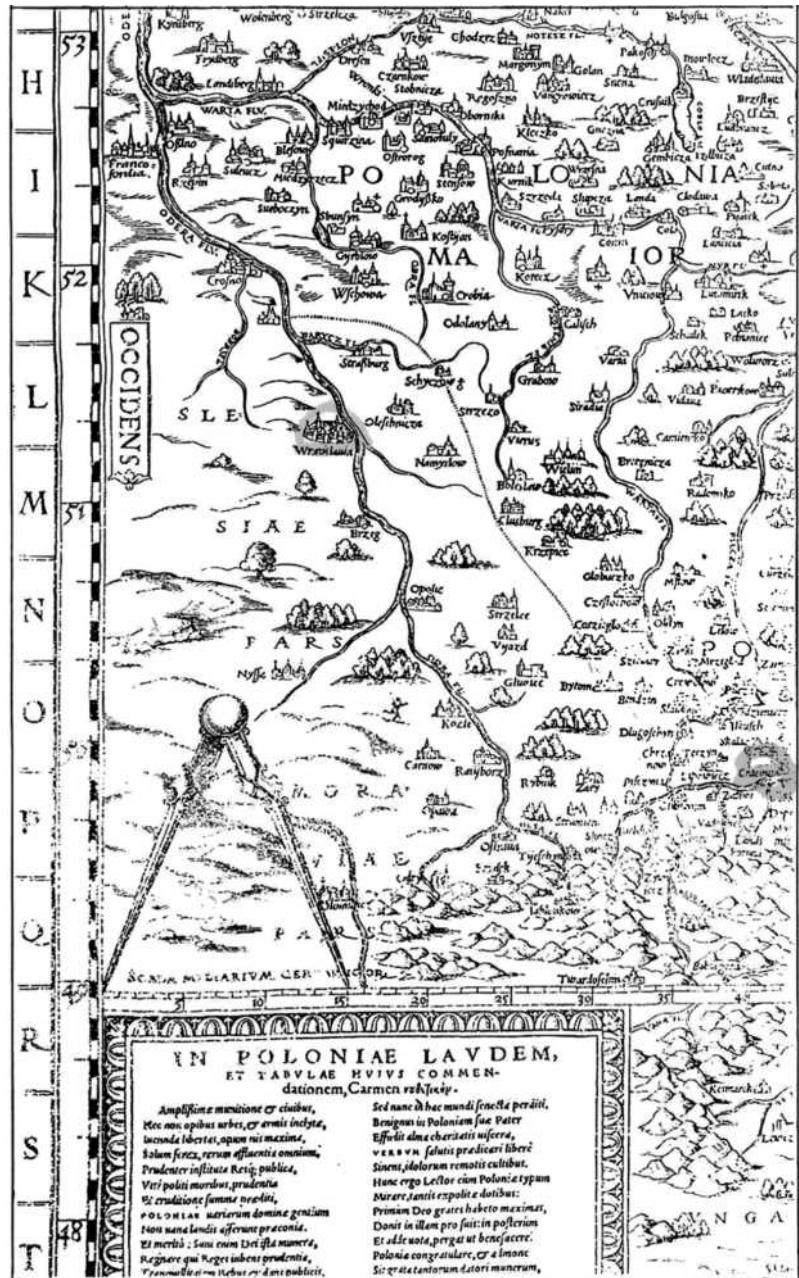


Figure 1. Silesia from a part of Venceslaus Grodecus' map 1558–1562 (1).

where the king's court and an increasing number of citizens created high demand for fish. In some estates, the income from pond farming was estimated at 50–60% of total income.

The fish caught were distinguished as white and black fish; pike and carp were mentioned separately (Fig. 4). The separate rearing of fish fry in small ponds, chiefly by peasants, points to the high level of fish culture at that time. The need for the development of technologies for pond construction and water leveling found response in one of the first technical books in Polish written by Olbrycht Strumieński and printed in Cracow in 1573 (Fig. 5).

In towns, fishers' guilds were organized to preserve the acquired rules of fishing in rivers and of fish trading. A Czech annalist, J. Dubravius, in 1600 highly appreciated

pond farming in Silesia, highlighting the abundance of inexpensive fish in comparison with Czech, Moravian, or Austrian (8,10).

Pond complexes also alleviated and prevented frequent freshets and floods in mountainous regions where forests disappeared owing to developing ferrous metallurgy. In periods of drought, pond waters were used to irrigate fields. Ponds were also built to reclaim marshy sites and to increase the area of cropland and meadows.

THE THIRD PERIOD

At the beginning of the seventeenth century, the fish farming economy collapsed, and this lasted to the middle of the nineteenth century. This was above all due to

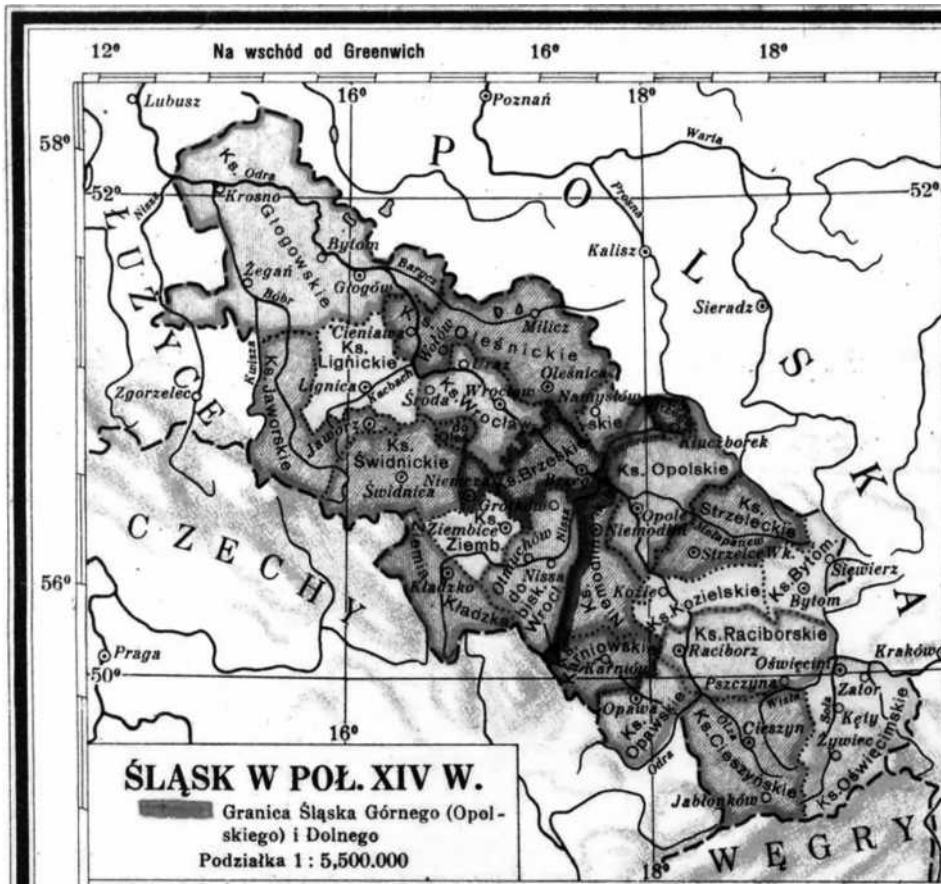


Figure 2. Partition of Silesia in the middle of fourteenth century. Semkowicz, W., *Szkolny Atlas Historyczny*. 1932, 2nd ed., 8, Książnica—Atlas, Lwów—Warszawa.

the Thirty Years' War (1618–1648) waged on a religious background. The war severely affected Upper Silesia. The behavior of foreign armies passing through the territory, for whose protection they had been called, had the most devastating effects. The majority of soldiers treated this province as enemy terrain causing enormous violence and the devastation made all inhabitants of numerous villages flee. The breakdown of agriculture due to depopulation created hunger and pestilence. Together with a very high contribution paid by the regent on behalf of the Austrian Empire, these events accounted for the poverty of the region (11). The religious repression was a further unfavorable stress on this country. The reconstruction of devastated ponds was impeded by the depopulation; however, the centers of large pond farms were renovated with the help of the vassal service. A decrease in the level of groundwater due to the development of coal mining in Upper Silesia dried up shallow peasant ponds (Fig. 6).

The strengthening of the feudal social system and the excessive burden of vassal service also affected peasant fish farming unfavorably. The primitive ineffective method of rearing all generations of fish in one pond returned (12,13). In Upper Silesia, the area of ponds decreased by 60% after the Thirty Years' War. A further reduction of about 20% occurred in the eighteenth century owing to the devastating Austrian–Prussian wars (1740–1742 and 1744–1745) (8). The high profit from cereal cultivation contributed to the decline of the fish-pond economy and further reduced the area of ponds in the

eighteenth century (12). However the area of ponds still reached 30,000 ha in Silesia, about 30% of the whole water surface area in the region at the turn of the nineteenth century (Fig. 7).

In the nineteenth century, the abolition of vassal service made the renewal of pond farming very difficult in Upper Silesia because the work was based chiefly on the serf system. In Lower Silesia, the area and number of ponds was reduced to a smaller degree; they were reconstructed by hired specialists, although the regulation and canalization of the River Odra by Prussian authorities decreased the groundwater level and contributed to the disappearance of numerous irrigated territories, also fishponds.

A probable reason for the breakdown of the fishpond culture might also have been the cold weather that prevailed from the midseventeenth to the midnineteenth centuries whose maximum was in the midseventeenth (4,14–16). The cooling, called the Little Ice Age, was also confirmed by decreasing carp sales and an increasing price of this fish species in the Plesnensis Principality from 1690–1810 (Fig. 8).

THE FOURTH PERIOD

A period of continuous development in pond carp culture began from the middle of the nineteenth century and was maintained till the present. It resulted from a fairly



Figure 3. Numerous fishponds in a part of the Upper Silesia. (from a Jonas Nigrini map 1725, copyright by Macierz Ziemi Cieszyńskiej, 2000).

Two groups of freshwater fish distinguished in the sixteennt century

"whitefish"	"blackfish"
Bream - <i>Abramis brama</i> (probably also <i>A. sapa</i> , <i>A. ballerus</i> , <i>Blicca bjoerkna</i>)	Tench - <i>Tinca tinca</i> Crucian carp - <i>Carassius carassius</i> Burbot - <i>Lota lota</i>
Roach - <i>Rutilus rutilus</i> (probably also <i>Scardinius erythrophthalmus</i>)	Loach - <i>Misgurnus fossilis</i> (probably also spined loach - <i>Cobitis taenia</i>)
Perch - <i>Perca fluviatilis</i> (probably also <i>Stizostedion lucioperca</i>)	Wels - <i>Silurus glanis</i> Carp - <i>Cyprinus carpio</i>
Gudgeon - <i>Gobio gobio</i>	
Ruffe - <i>Gymnocephalus cernua</i>	
Pike - <i>Esox lucius</i>	

Figure 4. Two groups of freshwater fish distinguished in the sixteennt century (elaborated by M. Kuczyński).

peaceful time at the end of the nineteenth and the beginning of the twentieth centuries. The expansion of numerous studies on carp physiology and diet played an important role, but the main impulse for the culture development was observations by Tomas Dubisz of

an accidental transfer of larvae during a freshet from spawning ponds to nursery ponds, previously plowed and sown with clover (Fig. 9) (18).

The observation caused him to introduce a new method of carp rearing, which ensured the abundance of food for

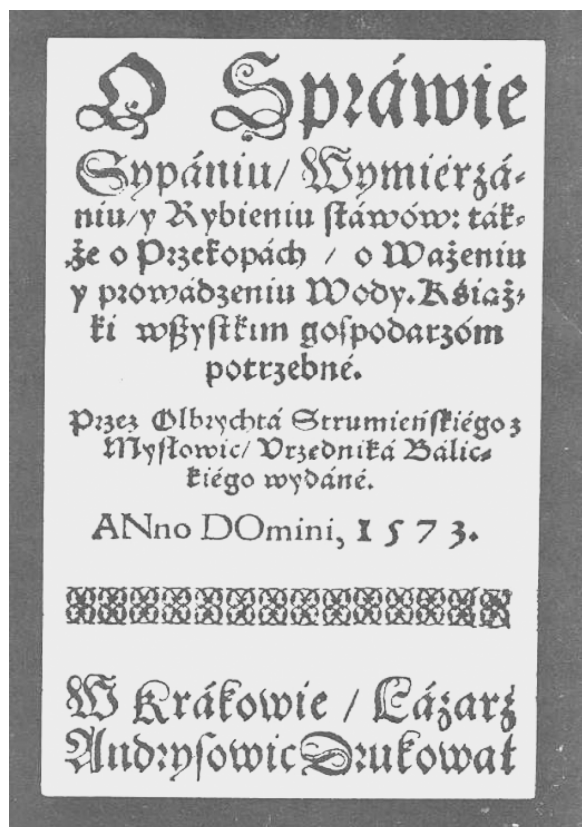


Figure 5. Cover of the book "About Managing, Pond Building and Stocking, Also About Digging, Water Levelling and Conducting, Book Useful for Every Farmer" by Olbrycht Strumieński, 1573 (9).

carp fry and shortened the rearing period of the market carp from 6–7 to 3 years. The effects of the fishpond culture also increased due to the warm period from the middle of nineteenth century (4,13). In Upper Silesia, most large fishponds were reconstructed by making them smaller and shallower; this created beneficial conditions for an increase in food resources for the carp and facilitated fish culture technology. However, during this time, large areas of ponds that vanished were turned into land ecosystems.

In the twentieth century, further steps in the intensification of fish culture consisted of the introduction of nitrogen–phosphorus fertilization in ponds. The application of this method in the 1950s increased yields from several kilograms to several hundred kilograms from 1 hectare of pond. A new intensified method of carp rearing introduced in fishery practice in the 1970s was based on greater stock densities and feeding fish pellets which covered the requirements of fish. By this method, yields rose to several thousand kilograms from 1 hectare. The polyculture of carp and herbivorous fish and the aeration of pond waters maintain proper environmental conditions (19).

In recent years, great attention was paid to the fishponds as ecosystems integrated with a catchment. Methods of the semi-intensive carp production, introduction of the polyculture with herbivorous fish species, and limited pond fertilization should ensure the sustainable functioning of carp ponds in river drainage. Moreover, these ecosystems enrich the river drainage in many ways (20).

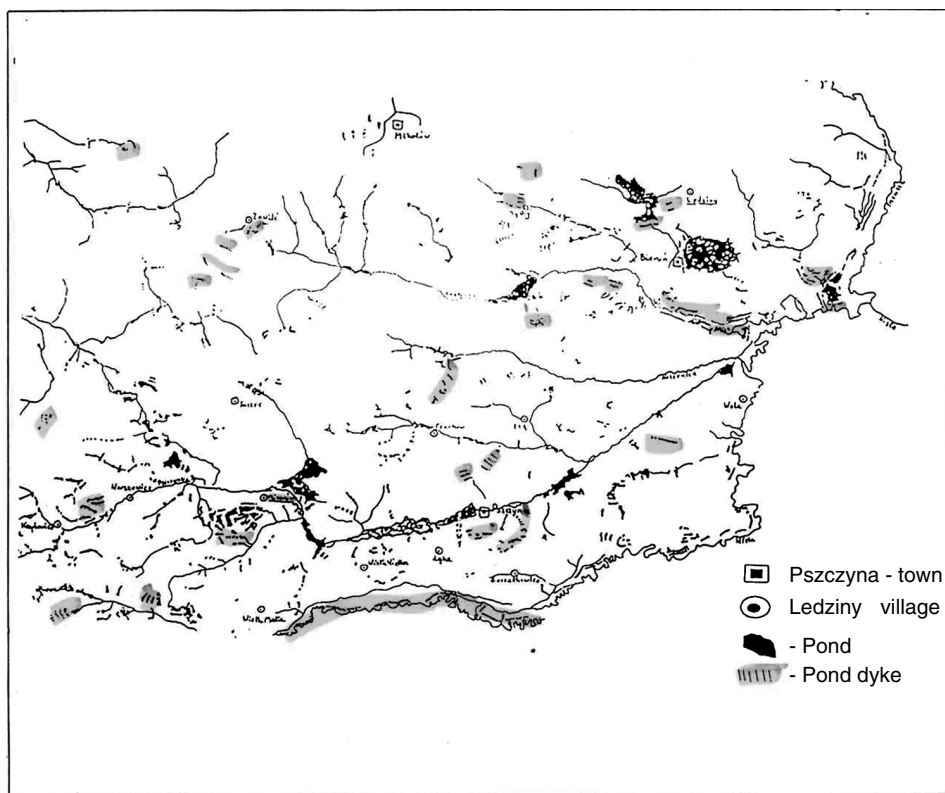


Figure 6. Numerous pond dikes in the Principatus Plessenensis in the second half of the eighteenth century (from Schlenger's map elaborated by Janczak, Nyrek, and Wiąrowski 1961) (14).

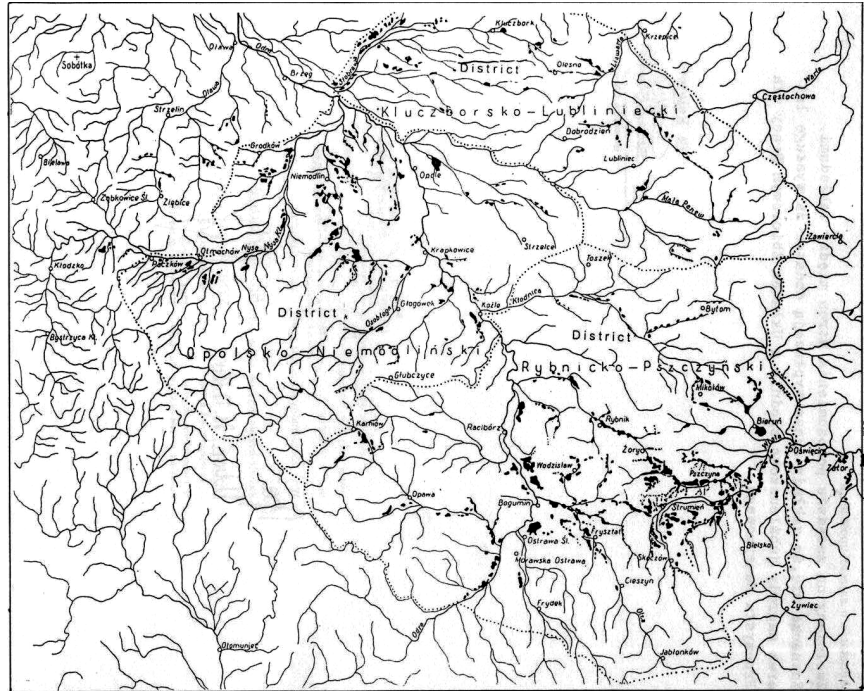


Figure 7. Water net and fishponds in Silesia in the first half of the eighteenth century (from Wieland-Schubbart's map 1736 elaborated by Nyrek (8).

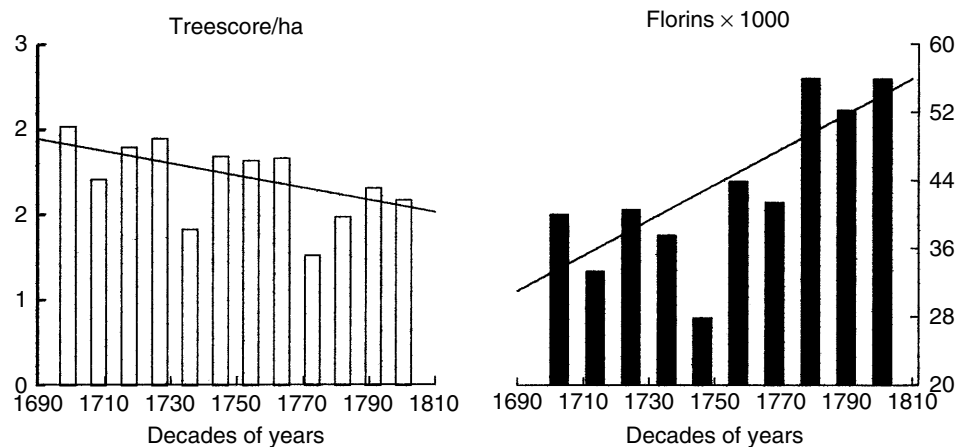


Figure 8. Average 10-year production and income from carp culture in ponds of Principatus Plessenensis between 1690 and 1810.

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Figure 9. The Thomas Dubisz brevet (17).

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FLOODS

STEVEN JENNINGS

EVE GRUNTFEST

(from *The Handbook of Weather, Climate, and Water:*

Atmospheric Chemistry, Hydrology, and Societal Impacts, Wiley 2003)

INTRODUCTION

The interface between humans and hydrologic features across Earth's surface has helped shape human culture. From the earliest agricultural, complex societies established along some of the great rivers of the world to the bustling seaports of today, humans have gained from the myriad advantages of living in proximity to water. Fertile soil, ease of transportation, and availability of resources (both materials and energy) have allowed for the development of complex material and intellectual cultures. The relationship between water and humans also brings a great deal of risk. Flooding is one of these risks. The impact of floods on humans has been evident from *Genesis* to tonight's evening news. Early Mesopotamian maps may have been drawn to facilitate the reestablishment of property lines after flooding. While the impacts of flooding on humans have been positive in the case of fertile floodplains that support much of the world's agricultural productivity, there is the potential for a great deal of negative impact (Brown, 1984; Clark et al., 1985). Losses of life and property have focused the efforts of scientists, engineers, and government agencies on the prediction, control, and mitigation of floods and flood damage.

In spite of efforts to deal with flooding problems, monetary losses continue to rise at an alarming rate. In Venezuela in December 1999, two weeks of heavy rain resulted on December 15th in flash floods laden with soil, vegetation, and debris. Damages of US\$3.2 billion, or 3.3% of the country's gross domestic product were reported. At least 20,000 people were killed. Generally, the number of lives lost due to flooding remains high. However, improvements in flood warnings particularly for major large-scale storms such as cyclones and typhoons have had dramatic effects. The severe 1991 cyclone in Bangladesh resulted in 140,000 dead and property losses of US\$2.0 billion. A cyclone of similar intensity in 1993 resulted only in the loss of 126 lives. The early warnings and cyclone shelters accounted for the major improvement (www.ndndr.org). China has also witnessed a reduction in the number of lives lost to floods. While 3000 people died in 1998 floods, the 1998 floods were as great as those of 1931 and 1954 where the loss of lives was 145,000 and 33,000, respectively (www.ndndr.org). In October 1998 hurricane Mitch was the worst in the eastern Caribbean since 1780 when a hurricane killed 22,000 people. The death toll from Mitch is reported as 11,000. More than 3 million people were left homeless or were severely affected (www.ncdc.noaa.gov/ol/reports/mitch/mitch.html).

Floods take a variety of forms with the interplay of several factors leading to the inundation of normally dry land.

Wohl (2000) identifies four primary challenges in reducing escalating flood damages. These are (1) estimating flood magnitude for a given recurrence interval, (2) accurately forecasting floods based on rapidly evolving weather conditions, (3) effectively operating flood-warning and evacuation procedures, and (4) establishing and enforcing land-zoning regulations. This chapter first discusses the contexts and causes of flooding, the first two points addressed by Wohl (2000). The second topic is the complex human responses to floods, points 3 and 4 of Wohl (2000). Many of these topics are illustrated with examples of floods.

DEFINITION OF FLOODS

Streams are linear water features that flow under the impetus of gravity. The amount of water contained in a stream is usually regulated by contributions of groundwater and surface runoff to the stream channel (Zaslavsky and Sinai, 1981; Knighton, 1998). Much of the time water in a stream flows within the confines of its channel. When inputs of water increase sufficiently, stream discharge leaves the stream channel and covers all or parts of the adjacent floodplain. Since the floodplain surface is usually a virtually flat surface and near the elevation of the stream channel, water can easily spread over the floodplain once water exceeds the elevation of the stream's banks. Most floods develop over a period of days or months as discharge increases gradually (Hirschboeck, 1987, 1988). Flash floods by contrast occur suddenly with little warning and are of short duration. Semiarid and arid areas are likely to experience flash floods (Reid and Frostick, 1987; Hassan, 1990). Flooding is not always associated directly with stream channels. Flooding occurs any time when water covers a surface that is normally not under water. Flooding can occur in coastal areas, low lying areas with poor drainage, or locations with inadequate urban drainage systems.

FACTORS THAT LEAD TO FLOODING

Floods have a multitude of causes. Some causes are related to what would be considered natural processes that would occur whether humans are present or not. Many causes have been affected by human activities. In some cases the severity of floods and the types of damage are a direct result of agriculture, urbanization, and the areas selected for development. In all cases flooding is related to increased discharge in stream channels.

Saturated Soil

Much of Earth's surface is covered by a weathered cover of regolith. Whether forming a true soil with well-developed horizons or a weakly developed detrital cover, the regolith is composed of a mix of mineral particles, organic fragments, and pore space. Commonly, much of the pore space is filled with air and to a lesser extent water. When large amounts of precipitation are received in a region, the pore space fills with water as the input of water from precipitation exceeds the output of water from the soil column to the water table. Decreases in

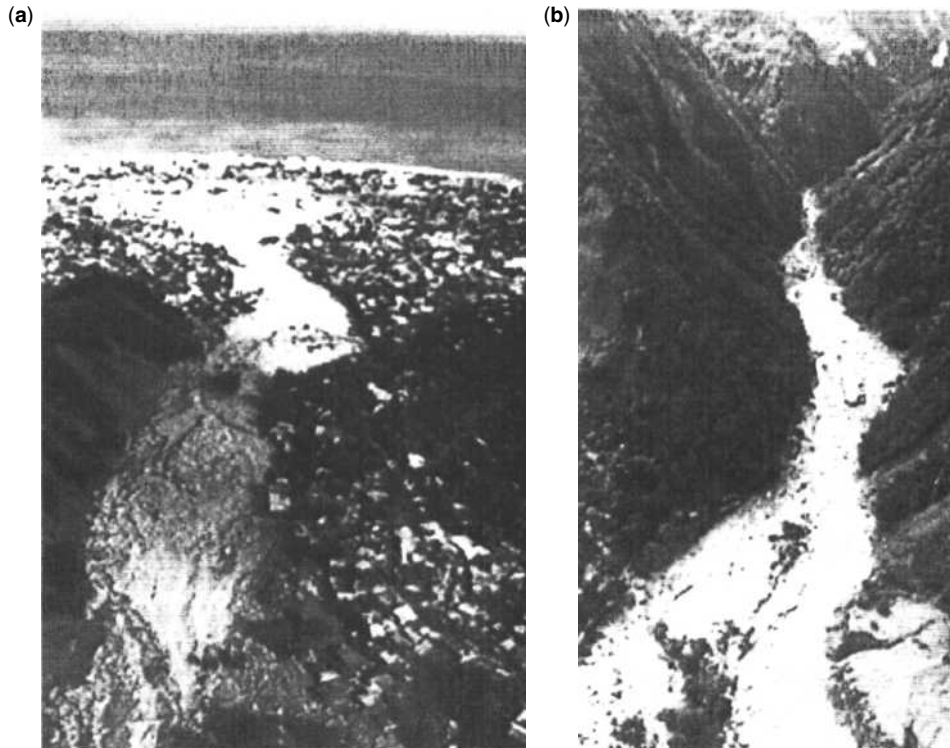


Figure 1. Quebrada San Julián upstream of Caraballeda showing evidence of recent debris flows and flash floods. Note the high slope angles, large numbers of debris flow scars, and abundance of new alluvium and colluvium in the channel bed and fan surface.

infiltration lead to increases in runoff. The lag time between the precipitation event and the arrival of water to stream channels decreases significantly when soil saturation occurs. As a result, peak discharge increases significantly and the likelihood of overbank flow is high (Smith and Ward, 1998). Spatially, soil saturation may occur over large-scale basins, which leads to flooding in large areas. The peak discharge flows downstream and becomes concentrated in higher order streams causing flooding. In many cases saturation follows a period of high amounts of precipitation over a prolonged time period, possibly weeks or months (Wolman and Gerson, 1978; Ward and Robinson, 1990).

Basin Characteristics

Surface characteristics influence infiltration and runoff rates (Roberts, 1989; Kuhnle et al., 1996). Impervious surfaces such as exposed bedrock or a paved road accelerate surface runoff, thus decreasing lag time between the precipitation event and entrance of water into a nearby channel. Urbanized areas, therefore, with large percentages of impervious surface such as roofs, streets, and parking lots coupled with an engineered drainage system designed to move water quickly to stream channels greatly increase the chances that some flooding will occur after a significant precipitation event (Wolman, 1967; Hammer, 1972; Roberts, 1989; Newson, 1992). Conversely, rural areas with large areas of soil, natural vegetation, and the potential for a faster infiltration rate are less likely to have significant flooding resulting from a single precipitation event. Removal of as much as half the forest cover and a decrease of marsh land along the Yangtze

River in China has led to increased flooding. Half a billion people, or 45% of China's total population, reside on the banks or floodplains of the Yangtze and the area produces about 42% of China's gross domestic product. In 1998, 79.6 million people in three Chinese provinces were affected by repeat flooding on the Yangtze. The floods killed more than 3000 people. Fourteen million people were evacuated and 21 million were made homeless (Weather.ou.edu/spark/AMON/v2_n3/News/DR_980819-China12.html).

Topography

Topography will influence the rate at which precipitation will be incorporated as stream discharge (Patton, 1988). Steep, rocky canyon walls have low infiltration rates as well as a great deal of potential gravitational energy that leads to the concentration of discharge during a short period of time (Strahler, 1964). Alluvial plains usually have a much longer lag time between a precipitation event and the introduction of runoff water into a stream channel. When land cover on steeper slopes is affected by perturbations such as wild fire or building-related oversteepening of slopes, the likelihood of mass movement events is greatly increased. These events are usually related to unstable regolith on steep slopes, which is susceptible to failure when sufficient precipitation is received. For example, see Fig. 1.

High Amounts of Precipitation

Flooding is created by the delivery of larger than normal amounts of runoff into stream channels (Smith and Ward, 1998, p. 67). Periods of above-average precipitation

lead to floods. In some cases seasonal variability leads to great fluctuations in stream discharge. Wet–dry subtropical or monsoonal climates with distinctive seasons of precipitation lead to fluctuations from dry stream channels to potential flooding events. These cyclical events are related to large-scale atmospheric circulation patterns that operate through an annual or longer period. In the midlatitudes, the annual migration of subtropical high pressures and the polar front lead to distinct precipitation patterns. In the tropics, monsoonal flow can lead to large precipitation events (Milne, 1986). On longer time scales El Niño and La Niña events are persistent over several years and can lead to wet or dry conditions over large areas of Earth's surface from the Equator to the midlatitudes (Waylen and Caviedes, 1987; Pearce, 1988; Ely et al., 1994).

Extended Wet Periods

In many cases flooding is caused by the reception of precipitation over an extended time period, on the order of weeks to months, that leads to the saturation of soils in a large-scale region (Rodda, 1970b; Smith and Ward, 1998). This saturation leads to increased runoff at a time when streams are at capacity (Ward and Robinson, 1990). Additional water introduced to stream channels cannot

be conveyed in the channel but is spread across the floodplain. Wet periods are related to synoptic conditions such as the position of the polar front that delivers cyclonic storms in quick succession. Poleward migration of subtropical air masses over continental areas such as the Mississippi River Basin help to supply large amounts of water to be precipitated by frontal activity. For example, see Fig. 2. In some locations rainfall may fall on snow-covered or frozen ground (Thomas and Lamke, 1962). These waters are unavailable to the hydrologic cycle as long as they remain in a solid form. In the case of the former, rainfall may accelerate the introduction of water into the stream network as snowmelt augments the precipitation already being received (Kattelman, 1990; Naef and Bezzola, 1990; Caine, 1995). The latter will greatly decrease the infiltration capacity of the soil causing most of the precipitation to quickly enter the stream network (Horton, 1933).

Decaying Tropical Cyclones

Some of the largest precipitation amounts received as the result of a single meteorological event have been associated with the movement of tropical cyclones (e.g., hurricanes, cyclones, and typhoons) poleward and over continents. These powerful cyclonic storms carry large

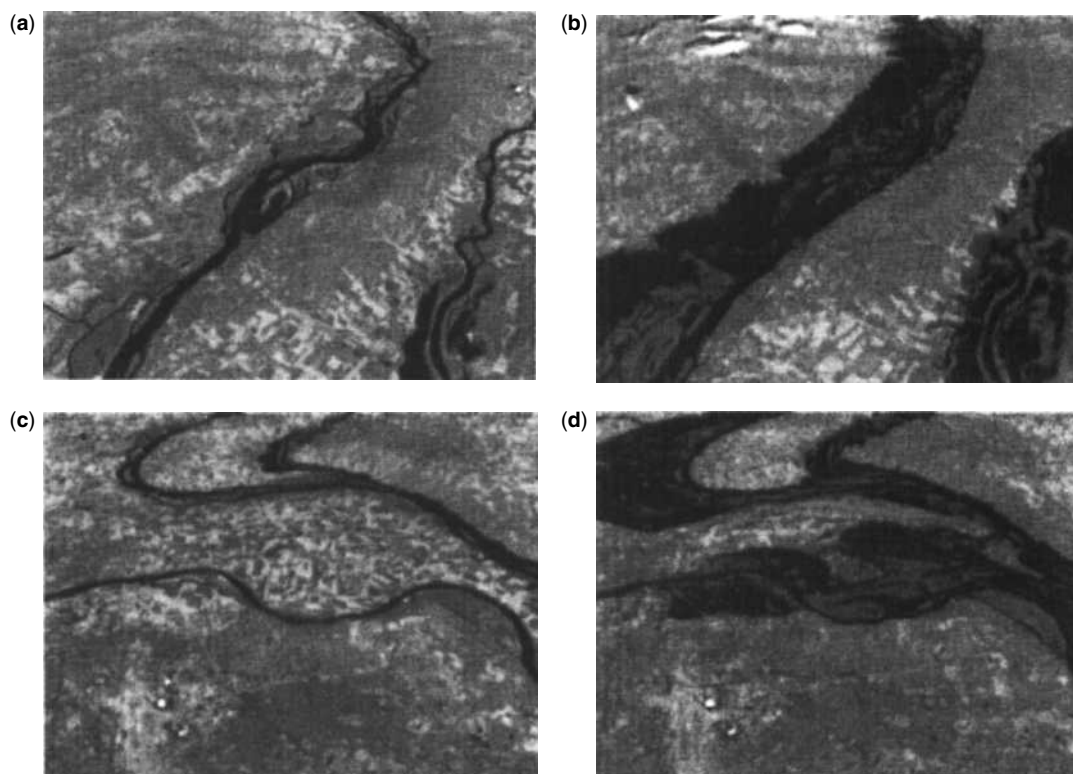


Figure 2. These scenes show various sections of the Mississippi River near St. Louis before and just after the 1993 floods, which peaked in late July/early August. The images show the area as seen by the LandSat Thematic Mapper (TM) instrument. The short-wave infrared (TM band 5), infrared (TM band 4), and visible green (TM band 2) channels are displayed in the images as red, green, and blue, respectively. In this combination, barren and/or recently cultivated land appears red to pink, vegetation appears green, water is dark blue, and artificial structures of concrete and asphalt appear dark gray or black. Reddish areas in the scenes during the flood show where water had started to recede, leaving barren land. See online version for color.



Figure 3. Water and sand washed inland to make travel difficult in North Topsail Island, North Carolina, after hurricane Fran.

amounts of warm moist air over land surfaces. While wind speeds associated with these storms decrease quickly after landfall, these decaying storms are capable of delivering precipitation over wide areas during a relatively short period of time, on the order of days to weeks. In some cases cyclonic storms associated with the polar front may exacerbate conditions by introducing a lifting mechanism that leads to increased condensation and precipitation. The relatively low-lying coastal plain of eastern North America is especially susceptible to damage from these types of storms (Bailey and Patterson, 1975; Hirschboeck, 1988). For example, see Fig. 3. In 1998 hurricane Mitch produced as much as 50 to 75 inches of precipitation in some areas of Central America. At least 11,000 deaths were associated with hurricane Mitch and more than 3 million people were left homeless or were severely affected (www.ncdc.noaa.gov/ol/reports/mitch/mitch.html).

Intense Thunderstorms

Thunderstorms are usually intense, short-lived storms that produce high winds, hail, and heavy rainfall. These storms can be caused by convection in moist tropical air masses over continental surfaces or fast-moving cold fronts that displace those moist air masses (Hirschboeck, 1987). When these storms develop over mountainous areas where the precipitation is concentrated by the topography the potential for large, catastrophic floods is great (Hall, 1981). For example, see Fig. 4. The eastern slope of the Rocky Mountains and the southwestern deserts of North America are common locations for the development of thunderstorms. As moist air encounters higher elevations in these locations, it is forced to rise. Unstable atmospheric conditions are created as mountain slopes heat and in turn heat the atmosphere. Adiabatic cooling causes condensation and the development of large cumulonimbus clouds that can reach the upper altitudes of the troposphere. Sometimes there is little movement associated with a thunderstorm or thunderstorm complex, with respect to the ground; heavy



Figure 4. Arizona flash flood, Wenden, Arizona. This community was flooded twice in late October 2000 when waters from Centennial Wash swept into the town. (Photo courtesy of U.S. Small Business Administration).

precipitation concentrated in a small geographical area can have catastrophic results.

Quick Snowmelt

The storage of water in the form of snow temporarily removes that water from the hydrologic cycle. In many cases this sequestration of water is short term. Snow accumulates during winter especially at higher elevations and latitudes. With the onset of warmer spring and summer conditions, snowmelt supplies water to streams. A typical early warming may mean that snowmelt may be accelerated with large amounts of runoff entering stream channels. Mountain ranges in mid-latitude coastal regions such as the Coast Ranges and Sierra Nevada of California receive a significant portion of their annual precipitation in the form of snow. It is possible for warm early spring rains to fall on the snowpack, causing much faster runoff than normal (Bolt et al., 1975; Church, 1988). Another source of snowmelt is the subsurface introduction of heat from volcanic activity. Large volcanoes can be high enough to support permanent snow and ice cover. High temperatures associated with volcanic activity lead to the instantaneous melting of snow and ice. The melt water is commonly mixed with pyroclastic debris to form lahars (Smith, 1996).

Failure of Flood Control Structures

A variety of humanly constructed structures are used in an effort to limit the extent and severity of flooding (Gregory, 1995). Dams and levees are common flood control structures designed to contain water within designated areas (Brookes, 1985, 1988). These structures can fail because of construction errors, poor design, and overtopping by water (Biswas and Chatterjee, 1971; Costa, 1988). Flood control structures can fail because of the failure of a key component. For example, a spillway that erodes away has the potential to lead to the catastrophic failure of the entire dam as the water cuts downward. Sound structures may fail when the water retained by the structure exceeds the height of the structure. Large precipitation events or the displacement of water in a

reservoir have the potential to send water flowing over the flood control structure (Kiersch, 1964). This may lead to the failure of the structure through erosion. Flooding may be exacerbated by these structures since a feature such as a levee tends to raise the stream level well above the floodplain. When a levee fails, a large amount of fluvial energy is concentrated through that break and a great deal of damage can occur near the break.

Cyclonic Storm Created Surges

In low-lying coastal locations a temporary increase in sea level associated with the approach and landfall of storms with significantly high winds and low central pressure can cause significant damage. Sea level rises in response to low pressure as it passes over ocean surfaces. Additionally, the upper portion of the water column is pushed into waves by the high winds. Storm surges can be more than 5 m above the normal high tide (Rappaport, 1994). In some areas such as bays coupled with low-lying deltas, like the Bay of Bengal and the mouth of the Ganges, where storm energy is concentrated, storm surges can reach high levels causing significant flooding (Frank and Husain, 1971; Murty and Neralla, 1992). Barrier islands are also susceptible to flooding by storm surges. Development on barrier islands along the southeastern coast of North America has led to rising property damage related to storms.

Mass Movement Events

A variety of mass movement events, while strictly not fluvial events, behave in a similar way to floods (Carson, 1976). The gravitationally fueled downhill movement of poorly consolidated regolith results from the introduction of meteoric water that adds weight and decreases hillslope cohesion. These events can do significant damage. Several types of mass movement events are composed of a larger percentage of sediments than a typical stream. Events such as mudflows, or lahars, commonly may approach the viscosity and velocity of streams. Valleys can be filled with fine-grained sediments as the deposits dewater following the initial surge of water and sediment. A variety of factors lead to mass movement events. The removal of plant cover by fire may expose soil surfaces so that infiltration rates may increase and lead to the accumulation of water along failure planes in the regolith. In areas with a subtropical wet-dry climate, such as the Mediterranean climate type, the burning of plant cover during the dry season and a subsequent wet season before the reestablishment of plant cover leads to mass wasting events (Rice et al., 1969; Campbell, 1975).

Human Responses to Flooding

There are no accurate estimates of the population in the world's floodplains. Even in the United States, only broad estimates are available, but the trends to increased vulnerability are clear. In 1955 U.S. floodplains had 10 million occupants. Thirty years later the number doubled to 20 million and by the mid-1990s about 12% of the national population lived in areas of periodic inundation. One sixth of the nation's floodplains are urbanized, and they contain more than 20,000 communities susceptible to

flooding. Half of these communities have been developed since the early 1970s (Burby, 1985; Montz and Gruntfest, 1986; Alexander, 1993).

Many of the people at risk do not understand the potential consequences of the hazards they face. In the United States, flood damages exceed \$2 billion annually. Only 20 to 30% of eligible structures are insured against flooding. Federal and state disaster assistance accounts for most of the difference. In the United States, almost two-thirds of the residential flood losses result from events that occur once every 1 to 10 years, even though the 100-year floodplain regulation is standard (Alexander, 1993).

In the United States, floods tend to be repetitive phenomena. From 1972 to 1979, 1900 communities were declared disaster areas by the federal government more than once, 351 were inundated at least three times, 46 at least four times and 4 at least five times. As of 1993, the United States was said to spend \$9 billion a year on flood control and \$300 million on flood forecasting (Alexander, 1993; Conrad, 1998).

Definitions of Structural and Nonstructural Measures

Adjustments to floods can be broadly classified into structural and nonstructural measures. Nonstructural approaches involve adjustment to human activity to accommodate the flood hazard (White, 1964; James, 1975; White, 1974) whereas structural methods are based on flood abatement or the protection of human settlement and activities against the ravages of inundation.

Structural change involves modification to the built environment to minimize or eliminate flood damage directly or flood channel construction changes. For example, see Fig. 5. Structural measures are expensive. They may give the illusion of security but the record shows otherwise (Alexander, 1993). The security can be temporary. A flood can occur that is bigger than the design of the channel or levee, and changing priorities in flood control projects that require higher reservoir levels for recreation or water supply can diminish the efficacy of structural measures (Williams, 1998).

The failure of structural flood control works poses a significant threat to the lives of the people who live



Figure 5. Elevated home in West Virginia is a mitigation success story. Risk is greatly reduced to homes elevated before a flood.

downstream from a massive structural project such as a dam. More than 2000 people died in 1969 in Italy when the Vaiont Dam collapsed (Blaikie et al., 1994). Because of stringent engineering standards and a system of inspections, the United States has seen few major failures. However, many structures are at the end of their design lives of 50, 75, or 100 years.

Structural flood control is still the dominant idea in many parts of the world. Following the 1927 Mississippi River floods, when river levees collapsed and 200 people died, 700,000 were displaced, and more than 135,000 buildings were damaged (Moore and Moore, 1989), the Army Corps of Engineers did not abandon its dream of controlling all floods. Rather, it proposed building large dams upstream to reduce flood peaks to the capacity of the floodway between the levees (Williams, 1998).

Until the 1970s, most flood loss reduction efforts involved structural solutions. Although nonstructural measures were discussed as alternatives, they were rarely implemented. The shift from mostly structural to mixed structural/nonstructural measures began in the 1970s and continues today. The mix of adjustments varies for each situation. In Europe almost all measures that are taken have elements of combined structural and nonstructural measures. There has also been a move to be antistructural. Some dikes are being removed in favor of nonstructural or more environmentally sensitive techniques (Smith and Ward, 1998).

Nonstructural measures include floodproofing, land-use planning, soil bioengineering, warning systems, preflood mitigation efforts, and insurance. The simplest nonstructural measure is to accept the loss. Another nonstructural measure is to provide postflood relief. Protection of floodplain residents and users, and the supply of relief when they suffer damage, are forms of hidden subsidy (Alexander, 1993). This category includes aid provided by the Red Cross, voluntary organizations, and governmental agencies.

Nonstructural measures include flood insurance and land-use management, acquisition and relocation, floodproofing, preflood mitigation preparedness, outdoor warning systems, and soil bioengineering.

DISCUSSION OF NONSTRUCTURAL MEASURES

Flood Insurance, Floodplain Mapping, and Land-Use Ordinances

In 1968 the U.S. National Flood Insurance Program (NFIP) was launched. It made affordable insurance available to residents in flood-prone areas. In 1999 more than 18,000 communities belonged to the program. Participating local governments require developers to meet minimum standards designed to avoid damages that might be inflicted by a catastrophic 100-year flood. The program also requires property owners to purchase flood insurance to receive a federally insured mortgage (Myers, 1996). Flood insurance is a means for placing some of the burden of losses onto the people who take (or make) the risk, namely the floodplain users and residents (Alexander, 1993). Communities can participate in a Community

Rating System, established by the Federal Emergency Management Agency (FEMA), that allows them to show innovative strategies to reduce flood losses in return for lower insurance premiums for floodplain residents.

Before a community can participate in the flood insurance program, the flood hazard must be recognized, assessed, and mapped. These assessments include flood history, cost and types of past flood damages, maps of the limits of the 100-year flood (or other designated flood) on a topographic map, compilations of profiles and cross sections of the river to show the levels of past floods, and compilations of flood frequency curves and locally representative hydrographs.

FEMA works with the state and community governments to identify their flood hazard areas and publishes a Flood Hazard Boundary Map of those areas. When a community joins the NFIP, it must require permits for all construction or other development in these areas and ensure that the construction materials and methods used will minimize flood damage. However, there is not careful monitoring to be sure that reducing flood hazard in a particular area does not increase flood potential elsewhere. Often, the problems are just shifted to different locales. In return the federal government makes subsidized flood insurance available to those whose structures were in the flood hazard area prior to issuance of the flood maps. All others are eligible for flood insurance at actuarial rates. FEMA issues a Flood Insurance Rate Map after the Flood Insurance Study of risk zones and elevations has been prepared (<http://floodplain.org/Jan32.htm>).

Acquisition and Relocation

The most effective measure to reduce losses is to keep the floodplains free of development. However, in many river valleys in the world, it is too late for that option. One of the most promising strategies for reducing flood losses is the public acquisition of developed land susceptible to flooding (Conrad, 1998; www.fema.gov/mit/homsups.htm). The authorization for U.S. federal cost sharing for relocation is more than 30 years old. However, only recently have communities, tired by chronic flooding, taken advantage of funding packages and relocated. In one case, the entire town of Valmeyer, Illinois, was relocated. The town had a long history of floods. In 1943, 1944, and 1947 unusually high levels of the Mississippi caused flooding in the nearby bottomlands affecting Valmeyer. After the 1947 floods, the U.S. Army Corps of Engineers raised the levees protecting the reach of the flood-plain to 47 ft. On August 1, 1993, the flood overtopped the levees inundating Valmeyer, prompting its ultimate relocation. Since 1993 nearly 20,000 properties in 36 states and one territory have been bought out and over 25,000 families have moved from floodplains (<http://www.nwf.org/nwf/pubs/higherground/intro.html>).

Floodproofing

Floodproofing is a range of adjustments aimed at reducing flood damages to a structure or to the contents of buildings. There are three categories: (1) raising or moving the structure; (2) constructing barriers to stop floodwater from

entering a building; and (3) wet flood proofing (U.S. Army Corps of Engineers, 1997).

Detection and Response Warning Systems

New technological advances in stream and rain gage networks and the increased regional floodplain management efforts have led to the adoption of thousands of local flood-warning systems. Many are simple detection systems and do not provide any mechanism for alerting the population at risk. In the United States until the 1990s warning or detection systems were planned and administered primarily at the local level.

Since then, the federal government including the Bureau of Reclamation, the U.S. Army Corps of Engineers, the National Oceanic and Atmospheric Administration, and the Federal Emergency Management Agency have actively participated in the installation and maintenance of detection and warning systems. Many systems are still managed by regional or local entities, but the percentage of federal dollars has increased substantially. Standards have also been established to help make the systems more compatible across regions (U.S. Department of Commerce, 1997).

An automated integrated network of stream and rain gages is being used in more than 1000 communities in the United States to help provide lead time for floods. Most of the systems are developed through collaborative efforts of many agencies. These ALERT systems (automated local evaluation in real time) have performed many functions other than flood warning, including helping in water supply decision making, fire weather forecasting, pollution monitoring, and providing data for river recreationists (Gruntfest and Huber, 1991). The availability of real-time data on the Internet also has increased interest in these monitoring systems (Gruntfest and Weber, 1998). The State of Arizona is developing a network for flood warning throughout the state. More than 30 agencies and communities are working together on the comprehensive ALERT system (<http://www.alertsystems.org/saas/>).

Warning systems may be nothing more than "cheap payoffs of the rain gods." Too often communities install rain gage/stream gage monitoring systems without a plan for getting the warning message disseminated. A warning system is only necessary once poor land-use decisions have been made, allowing people to settle in harm's way. Many of the systems being built are not being adequately maintained to be reliable (Gruntfest and Huber, 1991; Parker and Fordham, 1996). Public education encouraging people to heed environmental cues is also being used. It is particularly difficult to provide adequate lead times for flash floods. Some communities do have drills to test the reliability and completeness of their systems to be sure the systems will operate when the conditions warrant.

As of 2001 a combination of factors increase the likelihood that automated detection systems may become more popular and more valuable. More powerful, less expensive computers, and World Wide Web access provide opportunities for inexpensive real-time weather data. While real-time stream and rain gage networks may be originally installed for flash flood forecasting,

many agencies and users find the data useful for alternative purposes.

Soil Bioengineering

Anchored plantings along stream banks serve as the basis for this technique. Soil bioengineering and biotechnical engineering are cost-effective and environmentally compatible ways to protect slopes against surficial erosion and shallow mass movement. These approaches provide alternatives to structural channel "improvements." They raise questions about the notion of why engineers ever considered that concrete-lined channels should be considered "improved" (Gray and Sotir, 1996). Generally, bioengineering solutions must also include a strategy to carry floodwaters away.

The bioengineering technique is gaining support throughout the United States and Europe. It is less expensive to install and less expensive to maintain as well. The broader adoption of soil bioengineering may radically alter floodplain management.

Combined Structural and Nonstructural Measures to Reduce Flood Losses

From the first attempts to reduce flood losses in the United States, structural measures were preferred for three main reasons: (1) their benefits appeared to be relatively easy to measure, (2) they did not require extensive and politically controversial land-use planning, and, (3) the federal cost-sharing agreements encouraged communities to select the most expensive engineering projects. These reasons were supported by a faith in the technology of structural measures to protect people and property from floods.

The record now shows that in spite of massive expenditures, flood losses have continued to rise. Since the 1960s, especially in the United States, there has been a call for a shift from primarily structural measures to control floods to nonstructural measures (Galloway, 1994; Larson, 1996; Williams, 1998). Land-use control is one of the most effective ways of reducing flood hazards. Statutes, ordinances, regulations, and compulsory purchases can be employed and relocation can be subsidized. A floodway left undeveloped through the city can become beautiful public open space.

CONCLUSION

Floods are generally caused by the combination of large amounts of precipitation and basin topography. For example, saturation of soils caused by large amounts of precipitation can lead to flooding. Urbanization of a drainage basin increases the amount of runoff reaching a channel and decreases the lag time between a precipitation event and peak flow. A variety of weather events lead to flooding, including extended wet periods, decaying tropical cyclones, intense thunderstorms, and quick snowmelt. In some cases humanly constructed structures designed to prevent flooding collapse, causing flooding or accentuating flooding. In low-lying coastal areas storm surges may cause significant flooding. Mass movement events are similar to flooding, although the proportion of sediments to water

is larger than an alluvial flood with the outcome just as disastrous.

Humans respond to flooding in a variety of ways. Broadly defined these fall into two categories, structural and nonstructural measures. Structural measures include dams and dikes. Through time the efficacy of structural features has been questioned, and there has been a shift from purely structural approaches to controlling floods to a mix of structural and nonstructural flood mitigation strategies. Nonstructural measures include flood insurance, floodplain mapping, and land-use ordinances, acquisition and relocation, floodproofing, detection and response warning systems, soil bioengineering, and combined structural and nonstructural measures to reduce flood losses. Some progress is being made in addressing the hazards associated with flooding. The reduction of flood impacts continues at great expense, but vulnerability will continue to rise as long as more people build in floodplains, increasing the risk of catastrophic floods. Even the best warnings will not eliminate the risks increasingly being taken around the globe.

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SELECTED WEB PAGES RELATED TO NONSTRUCTURAL MEASURES

- <http://www.alertsystems.org/saas/ALERT> User Group
- <http://FEMA.gov> U.S. Federal Emergency Management Agency
- <http://ceres.ca.ca.gov/>—State of California water resources agency
- <http://www.usace.army.mil/inet/functions/cw/cwfpms/fpms.htm> U.S. Army Corps of Engineers with emphasis on floodplain management activities
- <http://www.nwf.org/nwf/pubs/higherground/intro.html> National Wildlife Federation site for manuscript Higher Ground
- <http://member.aol.com/damsafety/homepage.htm> Association of State Dam Safety Officials
- <http://www.ci.fort-collins.co.us/csafety/oem/index.htm> Comprehensive emergency preparedness homepage from Ft Collins, Colorado an excellent reference.
- <http://web.uccs.edu/geogenvs/work/Eve/Beyond%20Flood%20Detection%20Final.html>
- www.ncdc.noaa.gov/ol/reports/mitch/mitch.html—NOAA Website about Hurricane Mitch

FLOOD CONTROL STRUCTURES

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INTRODUCTION

Flood control structures are necessary to diminish the effects of excess water in streams. They can be distinguished in structures that provoke a reduction of peak flows and in structures that divert floodwaters through flood bypasses. Sometimes levees can also be used to control flood events.

STRUCTURES ABLE TO REDUCE PEAK FLOWS

They can be divided into retarding basins and reservoir flood storage.

Retarding Basins

The purpose of a retarding basin is to prevent flooding by controlling peak flows from storms to a level that could be safely carried by the downstream system (1).

It consists of a dam placed immediately upstream from the reach to be protected, preferably in an area where a great water volume is obtained with a small dam (2). The outlet discharge is generally regulated by valves or gates. The effect of a storage reservoir can be described with the equation of continuity:

$$Q_{in}(t) - Q_{ou}(t) = \frac{dV}{dt} \quad (1)$$

where Q_{in} is the inlet discharge, Q_{ou} the outlet discharge, and V the stored volume of water. Generally, the outlet discharge is a function of the water depth in the reservoir. A schematic draw of a storage reservoir is shown in Fig. 1, whereas the effect of water discharge is illustrated in Fig. 2. It is evident by a reduction of the peak flow and a modification of the duration of the flood event.

The type of the outlet should be chosen with respect for the storage characteristics of the reservoir and the nature of the flood problems. An ungated sluiceway is more preferable than a spillway operating as a weir, even if the latter is necessary for emergency discharge of a flood exceeding the design magnitude of the outlets.

Reservoir Flood Storage

Where flood damage at a number of locations on a river can be significantly reduced by construction of one or more

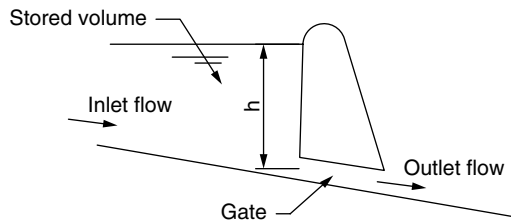


Figure 1. Schematic representation of a retarding basin.

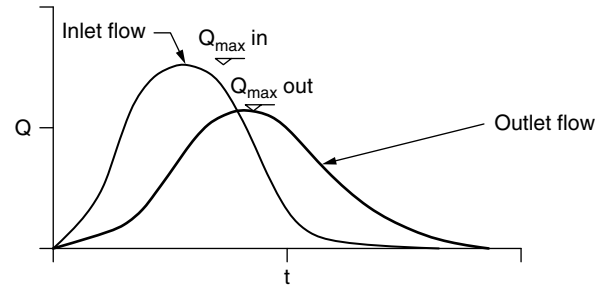


Figure 2. Effect of a retarding basin on flood discharge.

reservoirs, or where a reservoir site immediately upstream from one damage center provides more economical protection than local protection works, reservoir flood storages should be considered. These structures permit storage of a portion of the flood flow in a selected area, generally placed laterally to the river. The reservoir is fed with a lateral weir (Fig. 3). The release of the volume of the stored water is ensured by gates or valves and regulated in a way that the emptying happens after the flood has passed.

The effect of a storage reservoir is a diminution of peak flow without a modification of the duration of the flood (Fig. 4). The weir is dimensioned for the difference between the peak inflow (Q_{max} upstream) and the maximum discharge allowable downstream (Q_{max} downstream). Its height should be determined by evaluating the cost of the structure, because the taller it is, the longer it is. Besides, the height influences the point in which storage begins (point A of Fig. 4).

DIVERSION STRUCTURES

Flood bypasses ensure the diversion of floodwaters, which permits reduction of the flow passing in the original stream

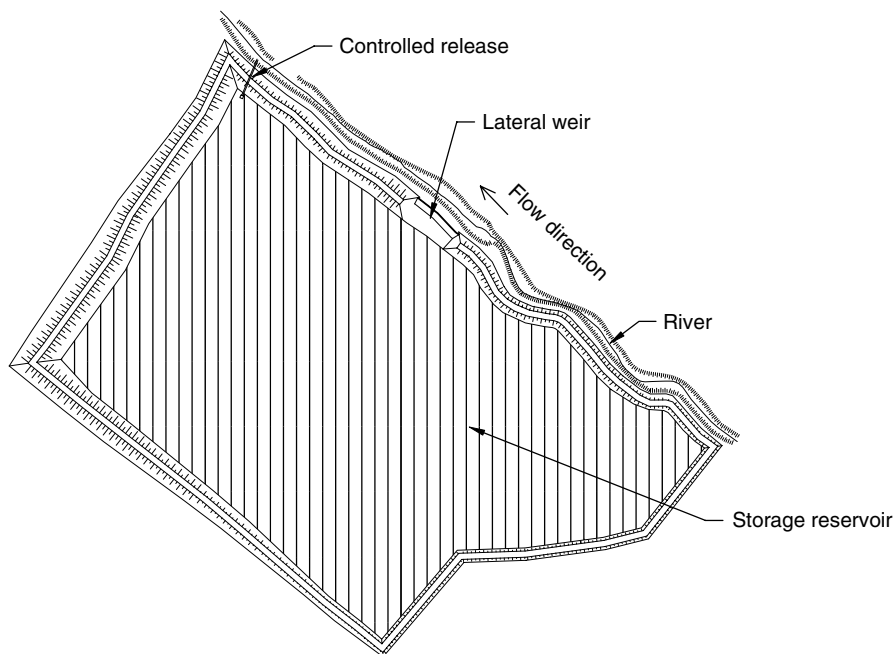


Figure 3. Storage reservoirs.

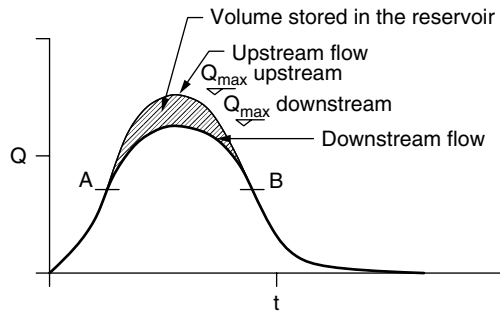


Figure 4. Effect of storage reservoir on flood.

to the allowable value (3). The diversion may return the water to the same channel at a point downstream or deliver it to another channel or different watershed. Besides, it not only conveys water in another reach, but also stores a substantial volume temporarily, thus serving as a large shallow reservoirs.

The diversion can be a second river working permanently (Fig. 5a) or if floodwater leaves the river at a controlled point, it works only during flood events (Fig. 5b). In the first case, modification to river morphology happens because there is a variation of the water tractive force in the original stream because of the diminution of the discharges. In the second case, the diversion remains dry for a long time and maintenance is necessary.

Admission of water to the bypass is achieved in different ways. In many cases, a low spot in the natural bank or a gap in the levee line is used. In some cases, a low section of the levee is created in a way that, when once

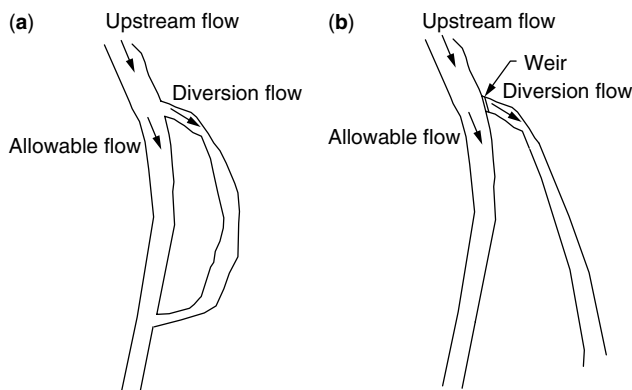


Figure 5. Diversion of flow.

overtopped, is washed out, and full discharge capacity is developed into the bypass. In other cases, a sill or a weir is used so that overflow occurs at a definite river stage. This last solution is suggested when overflow occurs frequently and the need for replacement of the levee section exists.

LEVEES

Levees represent a barrier preventing overtopping (4). Levees are longitudinal dams erected parallel to a river and can be earth dikes or masonry construction. In the last case, they are called flood walls.

Levees are more used because their cost is low and material is easily available. They are generally built of material excavated near the river and placed in layers and compacted (Fig. 6). To increase the impermeability of the levee, a core of clay or sheetpile septa can be used. A drain ditch or tile line along the back toe of the line is advisable and the back slope should be flat enough to contain the seepage line.

The width of the top should be enough to permit the movement of maintenance equipment with a minimum of about 3 m. Bank slopes are usually very flat because the low cost of the material, and they should be protected against erosion by sodding, planting of shrubs and trees, or use of riprap. Global stability should be verified during the design of the levee.

When dimension of leaves are not compatible with available area flood walls can be used to reduce the space. They are designed to withstand the hydrostatic pressure (including uplift) exerted by the water when at flood level. Sometimes flood wall is backed by an earth fill, and in case of low discharges, it works as a retaining wall.

CONCLUSION

Flood control structures ensure a reduction of risk and damage in case of a flood event. The commonly used structures for mitigation of flood damage have been investigated, pointing out the difference and the significant characteristics to proceed in a correct design.

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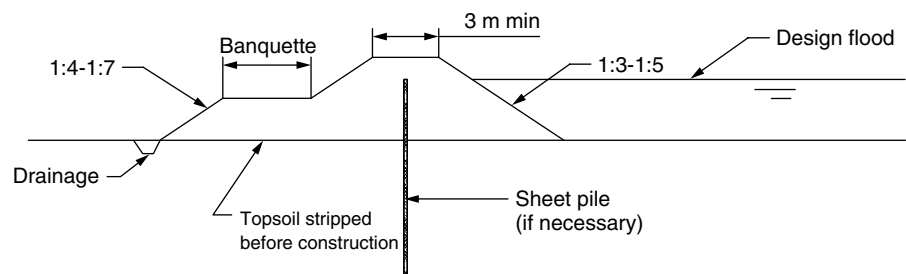


Figure 6. Typical levee cross section.

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FLOODS AS A NATURAL HAZARD

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The natural hazards suppose an important impact for human life and produce serious social effects and grave economic losses. Natural disasters constitute a constraint on the sustainable development, affecting its three basic mainstays: economics, social, and environmental.

In spite of the efforts made by the International Decade for Natural Hazard Reduction of the United Nations (1–3), the natural disasters in the world have experienced an increasing evolution during the last decades of the twentieth century, producing, at present, a mean of some 40,000 victims per year and mean economic losses of more than \$60 billion per year. The number of major natural disasters during the period between 1950 and 1999 has been multiplied by 4.4, and by more than 15 times with relation to the economic damages (Fig. 1 and Table 1) (4,5).

The economic losses because of natural disasters are increasing with an exponential trend, having tripled during the last decade of the 1990s, with a current evaluation of more than \$60 billion per year, as is shown in Fig. 2 (5).

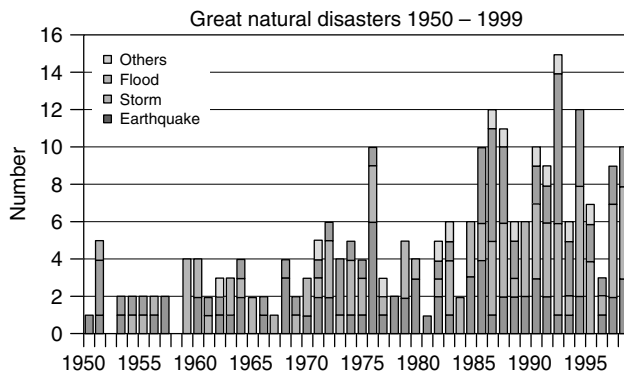


Figure 1. Number of natural disasters 1950–1999 (5).

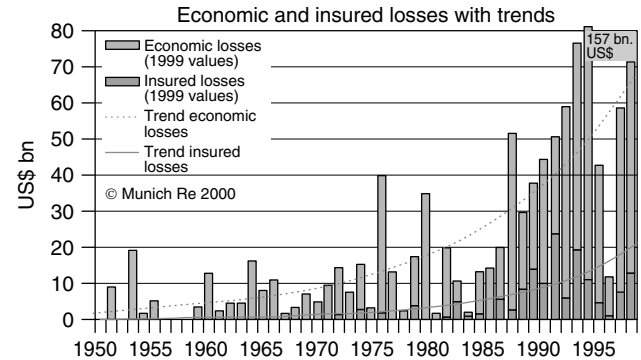


Figure 2. Natural disasters. Economic and insured losses with trends.

The greatest natural disasters of the last decades that have produced economic damages greater than \$10 billion are shown in Table 2. It can be observed that of the 19 disasters, 8 are big floods, such as those of 1993 in the United States; 2002 in Central Europe; 1991, 1996, and 1998 in China (Fig. 3); those of North Korea in 1995; and those of the year 1999 in Venezuela, with more than 30,000 victims.

The floods constitute the most important disaster among the natural hazards. Floods represent about 30% of the total number of natural disasters and economic damages, and almost 25% of the fatalities produced by the natural disasters (Fig. 4). The extensive and enhanced statistical data show that, in the period 1975–2001, some 95 significant floods per year have been produced, which have caused a mean of some 11,000 fatalities per year, and have affected some 150 M people per year, which signifies that in the last decade of the twentieth century, 25% of the world population has been affected by the floods (6).

In recent decades, the impact caused by floods has been very important, and Table 3 refers to the most catastrophic floods that have occurred in the last fifteen years.

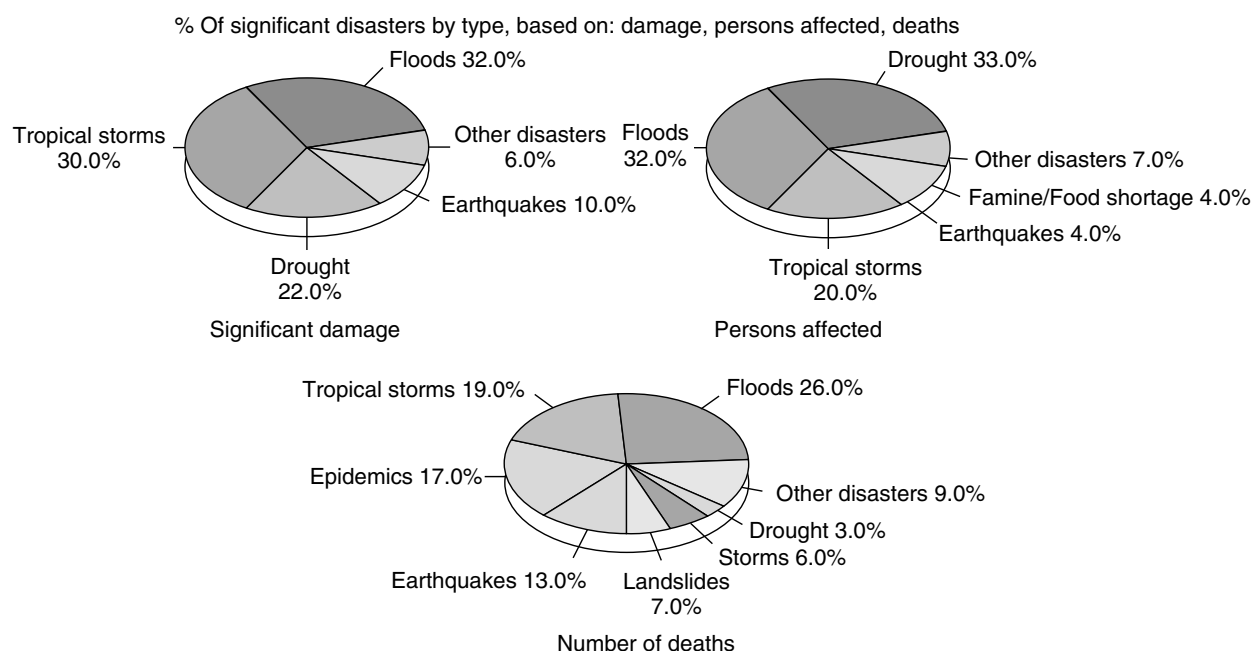
So then, experience shows that the impact caused by natural disasters and floods continues to increase progressively and, in many countries, constitutes a veritable restraint to sustainable economic development. For this, the UN decided in 1987 to create the International Decade for Natural Disaster Reduction (IDNDR) for the period 1990–2000 with the objective of reducing, by way of concerted international action, especially in the developing countries, the loss of lives, material damages, and the social and economic disorders caused by the natural

Table 1. Evolution of Number and Economic Losses of the Natural Disasters 1950–1999 (5)

Great Natural Disasters 1950–1999							
Decade Comparison Losses in US\$ Billion—1999 Values							
	Decade 1950–1959	Decade 1960–1969	Decade 1970–1979	Decade 1980–1989	Decade 1990–1999	Factor 80s:60s	Factor 90s:60s
Number	20	27	47	63	87	2.3	3.2
Economic Losses	39.6	71.1	127.8	198.6	608.5	2.8	8.6

Table 2. Major Natural Disasters in Relation with the Economic Damages

Country	Date	Disaster	Victims	Economic Damage (M \$)
Japan	17.1.1995	Earthquake	6,348	100,000
United States	17.1.1994	Earthquake	61	44,000
China	May-Sep.1998	* Floods	3,656	30,700
United States	23-27.8.1992	Hurricane Andrew	62	26,500
China	27.6-13.8.1996	* Floods	3,048	24,000
Europe	Aug. 2002	* Floods	230	18,500
United States	27.6-15.8.1993	* Flood	50	16,000
Venezuela	13-16.12.1999	* Flash floods, landslides	30,000	15,000
Korea (North)	24.7-18.8.1995	* Floods	68	15,000
China	May-Sep.1991	* Floods	3,074	15,000
EUROPE	25.1-1.3.1990	Winter storms	230	14,800
Taiwan	20.9.1999	Earthquake	2,474	14,000
Armenia	7.12.1988	Earthquake	25,000	14,000
United States	1.4-27.6.1988	Drought		13,000
Turkey	17.8.1999	Earthquake	17,200	12,000
Italy	23.11.1980	Earthquake	2,914	11,800
China	21.6-20.9.1993	* Floods	3,300	11,000
America	20-30.9.1998	Hurricane Georges	4,000	10,000
Japan	26-28.9.1991	Typhoon Mireille (N° 19)	62	10,000

**Figure 3.** Major disasters around the world 1963–1992. Percentage of significant disasters by type, based on damage, persons affected, deaths.

hazards. Among the essential elements of the activities of the IDNDR, the following points stand out (3,7):

1. Greater emphasis in planification and preventive measures.
2. Adoption of integrated actions (structural and nonstructural) for the reduction of the disasters.
3. Establishment of forecasting and alarm systems compatible with the technology and culture of the countries.

4. Development of a social conscience of the necessity of the reduction of the impacts.

In order to continue the activities developed by the IDNDR and promote international cooperation, the UN has implanted the International Strategy for Disaster Reduction (ISDR), resolution of the General Assembly 54/219 of the year 2000 (8). Among the principal objectives of the ISDR, the urgent need to develop further and make use of the existing scientific and technical knowledge to reduce the vulnerability to natural disasters,

Table 3. Most Important Catastrophic Floods 1990–2002

Country	Year	Victims	Economic Losses (US M\$)
China	Jul.–Aug. 1991	3,074	15,000
China	Jun.–Sep. 1993	3,300	11,000
USA	Jul.–Aug. 1993	38	15,600
Netherlands	Jan.–Feb. 1995	5	1,650
Norway	May.–Jun. 1995	1	240
Korea (North)	Aug.–Sep. 1996	68	15,000
Korea (South)	July 1996	99	600
China	Jun.–Aug. 1996	3,048	24,000
China	Jul.–Aug. 1998	4,150	30,000
Bangladesh	Aug.–Sep. 1998	1,655	13,000
Central America (Hurricane Mitch)	Oct. 1998	20,000	4,000
America (Hurricane Georges)	Sep. 1998	4,000	10,000
Venezuela	Dec. 1999	20,000	15,000
Mozambique	Feb–Mar 2000	929	1,000
Central Europe	Aug. 2002	230	19,000
Korea	Sep. 2002	150	800

**Figure 4.** China's 1998 flood.

stands out bearing in mind the particular needs of developing countries.

The ISDR has recently published a global review of disaster reduction initiatives “Living with Risk,” which is “the first comprehensive effort by the United Nations system to take stock of disaster reduction initiatives throughout

the world. This report discusses current disaster trends, assesses policies at mitigating the impact of disasters, and offers examples of successful initiatives. It also recommends that flood risk reduction be integrated into sustainable development at all levels—global, national and local” (6).

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FLOOD SOURCE MAPPING IN WATERSHEDS

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INTRODUCTION

Flood control projects are designed to protect people and properties against floods' adverse effects. No unique

methodology exists, however, to identify flood source areas and to evaluate their contribution to the flood characteristics at any downstream point of interest. The number of nested hydrometric stations and the amount of available flood data measured simultaneously at the stations are usually insufficient for ranking flood source areas. This constraint leads us to take advantage of hydrologic simulation models.

Lumped modeling approach in flood studies precludes spatial prioritization with respect to downstream flood response. Thus, semi- or fully distributed modeling approaches are appropriate for flood control studies in large watersheds. However, the main issue is how to identify and prioritize watershed flood source areas based on their contribution to the flood response at the main outlet. The proposed methodology is also useful to determine source erosion and pollution areas.

UNIT FLOOD RESPONSE APPROACH

In this section, the approach proposed by Saghafian and Khosroshahi (1) for flood source mapping is briefly described. This approach enables identification and ranking of flood source areas within a watershed. Although the approach can be extended to a fully distributed discretization scheme, we consider delineated subwatersheds to represent homogeneous unit areas. Unit areas that produce the largest contribution to the flood peak at a given point subject to flooding (usually the outlet of the watershed draining the area of interest) are given highest rank among all source area units. One must note that ranking of subwatershed units based on their contribution to the discharge at the main outlet is not necessarily equivalent to the ranking based solely on the magnitude of peak discharge produced at the outlet of each unit. Another issue is that the nearest or largest

subwatershed unit may not cause the largest impact on the outlet flood peak. These issues are studied later when a case study is presented. Particular emphasis will be put on the routing in the stream network that connects subwatershed areas because storage is a key factor in how flood source areas influence the outlet flood response. It is expected that the effect of stream routing is more pronounced for more frequent floods.

The flood source mapping procedure starts with collection of watershed data and historic rainfall-runoff data recorded at rain and stream gauges located inside or outside the watershed. A semidistributed rainfall-runoff model with stream routing capability is selected, and the model is properly calibrated and validated against the flood data of at least one hydrometric station located near the outlet of the watershed. Then the model is setup for the watershed of interest, if different from the watersheds bounded at hydrometric stations for which the model was calibrated.

Subwatersheds' flood hydrographs are simulated for a selected design storm, and flood routing is performed in the channel network connecting the outlets of subwatersheds to the watershed's main outlet. The routing results in a base outlet hydrograph. Then, in successive runs, each subwatershed unit is singly removed, and the resultant hydrographs at the main outlet are simulated (Fig. 1). The changes in flood peak of the generated hydrographs relative to the peak of the base hydrograph are computed. The procedure may be repeated for other design storms, and the sensitivity of the results could be examined.

We may now define a flood index to measure unit area contribution to the flood peak at the main watershed outlet:

$$FI_k = \frac{Q_{p,all} - Q_{p,all-k}}{Q_{p,all}} \times 100 \quad (1)$$

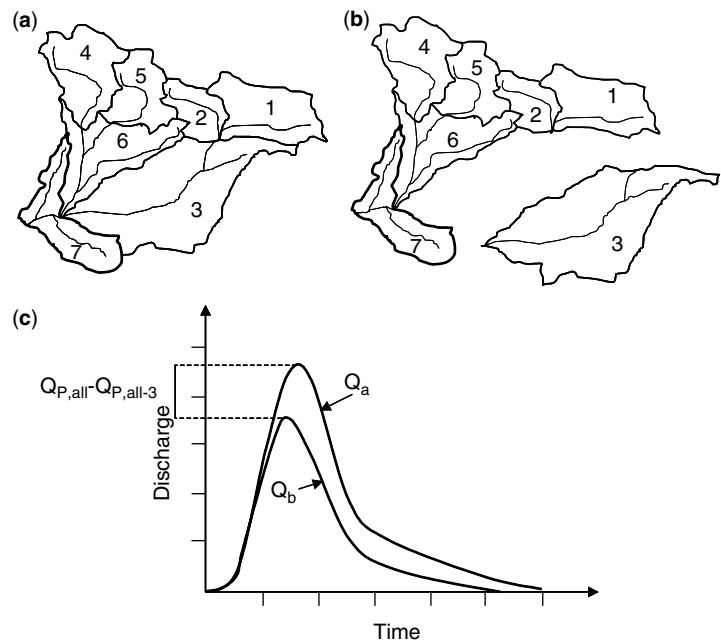


Figure 1. Unit flood response application on a watershed: (a) subwatershed units, (b) unit 3 removed, and (c) simulated flood hydrographs corresponding to cases (a) and (b).

where FI is the gross flood index of the k th subwatershed (in percent), $Q_{p,all}$ is the peak discharge of the base hydrograph (in m^3/s), $Q_{p,all-k}$ is the peak outlet discharge when the k th subwatershed is removed (in m^3/s), and A_k is area of the k th subwatershed (in km^2).

A different flood index can also be expressed in terms of contribution per unit area of subwatersheds as follows:

$$f_k = \frac{Q_{p,all} - Q_{p,all-k}}{A_k} \quad (2)$$

where f_k is the flood index of the k th subwatershed based on unit subwatershed area (in $m^3/s/km^2$).

The approach outlined above is entitled “unit flood response,” abbreviated by UFR, based on its similarity to the unit response (UR) approach in groundwater management. In the UR approach, a groundwater model simulates the influence of a unit change in sink/source rates at selected well locations on design variables, such as hydraulic head or velocity, at specified locations (2). Suitable groundwater quantity/quality management scenarios may be studied based on the UR application. Although nonlinearity can be preserved by the model used in the unit flood response approach, the UR in groundwater studies assumes linearity by superimposing the individual unit responses at given locations.

A CASE STUDY

Damavand watershed with an area of $758 km^2$ lies at $51^\circ, 46', 40''$ to $52^\circ, 12', 05''$ E longitude and $35^\circ, 32', 48''$ to $35^\circ, 51', 39''$ N latitude. High and low elevations in this relatively steep watershed are 4003 m and 1250 m above sea level, respectively. Rangeland, mostly in poor condition because of overgrazing, is the dominant vegetation cover. Agricultural areas are concentrated in the middle and outlet parts of the watershed. Average annual precipitation is reported as 443 mm. Although most precipitation occurs in the winter and early spring, severe floods in the region are often caused by high-intensity summer storms. The region is generally categorized as a flood prone area. As a result of limitations imposed by gradual availability of funds, one of the primary aims of flood control studies in this watershed is to identify and rank areas with higher impact on the outlet flood peak.

Boundary of Damavand watershed and its seven subwatersheds along with the contour map were digitized

in a geographic information system (GIS). Maps of digital elevation model (DEM) and land slope were generated by the GIS. Figure 2 shows subwatershed boundaries on the DEM background. Table 1 summarizes some of the physiographic characteristics of the subwatersheds. A land-use map was produced through processing a Landsat satellite image of the area and field inspection. SCS curve number layer was generated by overlaying maps of land use and hydrologic soil groups (Fig. 3). Then, average subwatersheds' CN values were determined (see Table 1).

HEC-HMS hydrologic model (3) is applied in this case study. Specifically, SCS unit hydrograph method and Muskingum technique were applied to simulate rainfall-runoff transformation at subwatershed scale and stream routing, respectively. Rainfall intensity corresponding to the design storm of 50-year return period with duration equal to the watershed concentration time was drawn from the intensity-duration-frequency (IDF) curve of a local station. Average rainfall temporal pattern was developed at the same station, but no particular spatial pattern was considered in this case study.

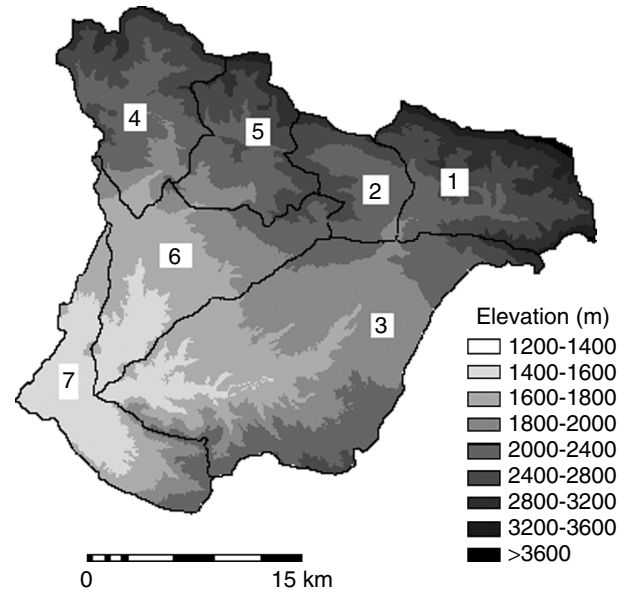


Figure 2. Subwatersheds of Damavand watershed on digital elevation model background.

Table 1. Damavand Subwatersheds Characteristics

Subwatershed Unit	Area (km^2)	Land Slope (%)	Max Stream Length (km)	Distance to Outlet (km)	Stream Slope (%)	Curve Number II
[1]	[2]	[3]	[4]	[5]	[6]	[7]
1	97	47	16.8	32.1	10.5	81
2	46	27	12.3	32.1	5.3	71
3	253	19	36.2	4.8	5.1	78
4	96	37	19.3	18.6	8.2	80
5	70	35	14.8	20.7	6.7	80
6	112	12	24.9	4.8	6.7	78
7	84	13	16.4	0	6.0	86

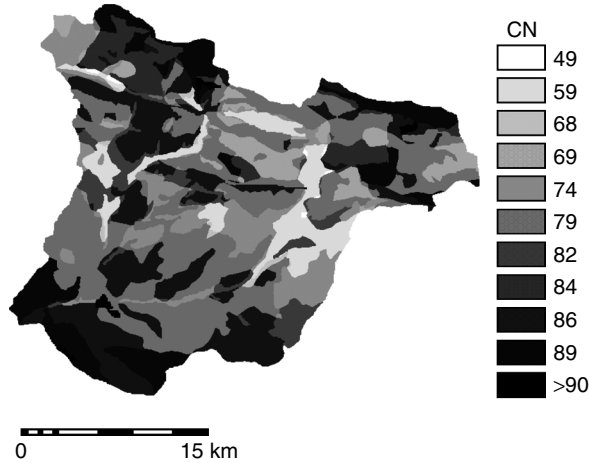


Figure 3. Map of curve number.

FLOOD INDEX ANALYSIS

HEC-HMS model was calibrated and validated for a number of rainfall-runoff events recorded at the outlet of the watershed. Calibration parameters were SCS initial loss and lag time. The 50-yr, 12-hr storm was simulated with peak discharge at the outlet of each subwatershed shown in Column 2 of Table 2. A peak discharge of 371.9 m³/s was produced at the watershed outlet. The simulated outlet peak discharges after removing each subwatershed are reported in Column 4 of Table 2. For example, the peaks corresponding to cases with and without subwatershed 3 are 371.9 m³/s and 270.0 m³/s, respectively. Therefore, the peak is reduced by 101.9 m³/s when subwatershed 3 is completely removed. Subwatershed 3's *FI* and *fi* flood indices are equal to $101.9/371.9 = 27.4\%$ (Column 5, Table 2) and $101.9/253 = 0.40$ m³/s/km² (Column 7, Table 2), respectively. The priorities based on the absolute peak discharge contributions (*FI*) and per unit subwatershed area peak discharge contributions (*fi*) are respectively listed in Columns 6 and 8 of Table 2.

The overall mixed ranking of subwatersheds in Columns 3, 6, and 8 in Table 2 highlights the complex interdependent effect of subwatershed area, subwatershed characteristics, channel routing, and the location of flood source areas within a relatively large watershed. It

is noted that the nearest (to the outlet) or largest and the most distant or smallest subwatersheds do not necessarily generate the highest and lowest contribution to the flood peak at the outlet, respectively. Similarly, subwatersheds producing the highest and lowest discharges at their own outlet may not rank first and last in *FI* or *fi*. Subwatersheds' absolute and specific discharges are compared in (Fig. 4), whereas *FI*'s and *fi*'s are shown in (Fig. 5).

One can observe, from (Fig. 5), that the rankings of absolute discharge contribution (*FI*) generally differ from those of per unit area discharge contribution (*fi*). Subwatersheds 5 and 4 rank first and second in the unit area discharge contribution, whereas subwatershed 2 attains the lowest rank. Subwatershed 3, with the highest ranking in the absolute discharge contribution (*FI*), ranks

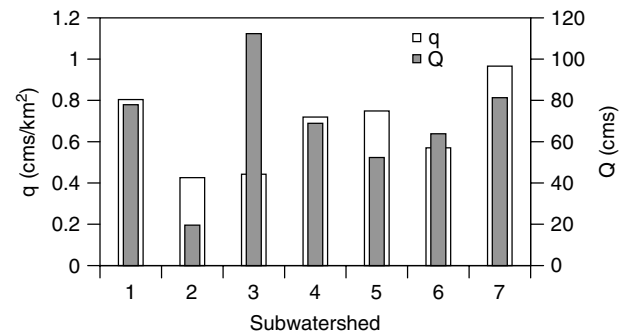


Figure 4. Comparison of absolute and specific subwatershed peak discharges.

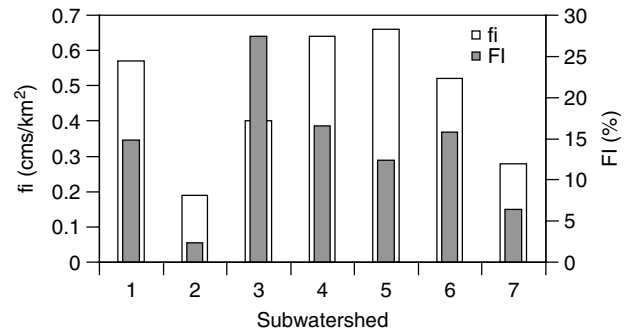


Figure 5. Comparison of subwatersheds' *FI* and *fi* flood indices.

Table 2. Simulated 50-yr Peak Discharges and the Results of UFR Application

Subwatershed	Subwatershed Peak Discharge (m ³ /s)	Priority Based on Subwatershed Peak Discharge	Outlet Peak Discharge Without Subwatershed (m ³ /s)	Flood Index <i>FI</i> (%)	Priority Based on <i>FI</i>	Flood Index <i>fi</i> (cms/km ²)	Priority Based on <i>fi</i>
[1]	[2]	[3]	[4]	[5]	[6]	[7]	[8]
1	77.9	3	316.8	14.8	4	0.57	3
2	19.6	7	363.0	2.4	7	0.19	7
3	112.2	1	270.0	27.4	1	0.40	5
4	69.1	4	310.0	16.6	2	0.64	2
5	52.3	6	325.8	12.4	5	0.66	1
6	63.7	5	313.3	15.8	3	0.52	4
7	81.3	2	348.1	6.4	6	0.28	6

Table 3. Flood Index (f_i) Matrix for Pairs of Subwatershed.

Subwatershed	1	2	3	4	5	6	7
1	0.568	0.439	0.426	0.593	0.596	0.558	0.472
2	0.439	0.193	0.366	0.499	0.474	0.436	0.273
3	0.426	0.366	0.403	0.456	0.448	0.450	0.396
4	0.593	0.499	0.456	0.645	0.649	0.585	0.481
5	0.596	0.474	0.448	0.649	0.659	0.580	0.456
6	0.558	0.436	0.450	0.585	0.580	0.523	0.399
7	0.472	0.273	0.396	0.481	0.456	0.399	0.283

fifth in the unit area contribution (f_i). Use of flood index (f_i) is recommended in the analysis of spatial flood source prioritization required for optimum planning flood control measures at subwatershed scale.

The effect of spatial aggregation of unit source areas could also be examined by removing a pair of subwatersheds at each simulation run. The flood index matrix is shown in Table 3. The f_i values in nondiagonal cells, say j th row and k th column, correspond to removing the j th and k th subwatershed simultaneously. When $j = k$ in diagonal cells, the results correspond to the case when a single subwatershed j has been removed (as already reported in Table 2). Clearly, the combination of subwatersheds 4 and 5 accumulated the highest impact on outlet flood peak. Conversely, the combination of subwatersheds 7 and 2 indicates the lowest impact.

SUMMARY

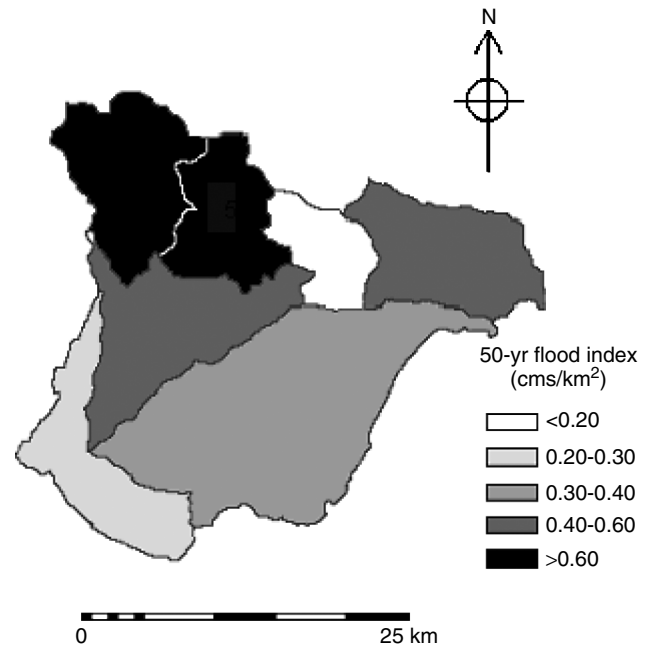
A technique, abbreviated by UFR, was described in this chapter for mapping flood source areas within a watershed. Subsequent elimination of subwatershed units in the rainfall-runoff modeling process allows separation of the effect of individual watershed area. Thus, contribution of each subwatershed unit may be quantified and its priority determined. The technique can be effectively applied in flood source area mapping at the desired spatial scale.

The UFR technique was applied in a case study, and the map of a 50-yr flood index was prepared (Fig. 6). Based on the results of this case study, it may be concluded that because of the complex nature of flood hydrograph propagation within the stream network, use of the UFR approach can be considered as the optimal way for prioritization of flood source areas.

Another issue is that implementing flood abatement measures in subwatersheds with low flood index may actually result in delaying peak arrival of such areas and could cause a more synchronized response with other subwatersheds, which is the case with subwatershed 7 in the presented case study.

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**Figure 6.** Map of 50-yr flood source units.

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URBAN FLOODING

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Flooding in urban areas is an inevitable problem in many cities, and it causes huge costs to the society in structural and nonstructural damage. The problems that arise from urban flooding range from minor ones such as water getting into the basement of a few houses, to major incidents, where large parts of cities are inundated for several days. Most modern cities in the industrialized part of the world usually experience smaller scale local problems due mainly to insufficient capacity in their sewer systems during heavy rainstorms. Cities in other regions, including those in South/Southeast Asia, often have more severe problems because of much heavier local

rainfall and lower drainage standards. This situation is continuously getting worse because many cities in the developing countries are growing rapidly these days but do not have the necessary funds to extend and rehabilitate their existing drainage systems.

CAUSES OF URBAN FLOODING

Urban flooding may be caused by a river or by local rainfall over a city. *River-induced flooding* is flooding of a city due to high water levels in rivers adjacent to the city. River water flows to lower lying areas, and floods them. *Rainfall-induced flooding* is flooding of a city due to local rainfall in the built-up areas of the city. Examples of rainfall-induced flooding in Bangkok can be seen in Figs. 2 and 3. This may occur several times a year on various scales. Mainly, inadequate existing drainage paths and their improper operation and maintenance cause these floods.

An interaction often exists between the drainage capacity of an urban drainage system and the water level in a recipient. For example, Bangkok was flooded from local rainfall during October 2002 due to reduced drainage



Figure 1. Flooding in a suburb of Dhaka City, 1996. (Photo by Terry Van Kalken).



Figure 2. Flooding in Bangkok during a minor premonsoon rain in April 2002. (Photo by Dr. Ole Mark).



Figure 3. Flooding in Bangkok, 2002. (Photo by Mr. Ebbe Worm).

capacity from the city to the river. The condition in the river was caused by a combination of very high runoff from the upstream river basin and a spring tide in the Gulf of Thailand, resulting in a very high water level in the river and hence drastically reduced drainage capacity from Bangkok to the river. No drainage by gravity could take place, and only drainage to the river by pumping was feasible.

Individual streets in a city may flood due to numerous causes (1). Frequently, rainfall runoff starts as an overland flow on the street before entering the underground pipe system through catch pits. The water in the pipe system is then conveyed downstream but may return to the surface if the capacity of the pipes is exceeded. The duration of flooding on the street depends on the ground elevation, the intake capacity of the catch pits, and the capacity of the underground pipes. If the intake capacity of the catch pits is limited, large volumes of runoff will be transported along the surface during large rainstorms, even if there is sufficient capacity in the underground sewer network. Insufficient drainage capacity locally in the sewer system may then, in turn, cause surface flooding, even if the catch pit intake capacity is adequate.

THE EXTENT OF URBAN FLOODING

In Asian cities subject to regular flooding, people seem to have adjusted their daily lives to smaller flooding. However, major incidents have grave consequences. Jakarta, Indonesia, was flooded from an overnight rain during February 2002. During that flood, five people were killed and approximate 200,000 people were forced from their homes. Another urban flood occurred in Bangkok, Thailand, in 1983, which inundated the city for 6 months. The flood caused damage of approximately US\$149 million. In 2000, an urban flood in Mumbai, India, claimed at least 18 lives, made hundreds of slum dwellers homeless, and plunged large areas of the city into darkness. In Dhaka City, Bangladesh, moderate rain can inundate parts of the city for several days. The situation was highlighted in

September 1996, when the residents of Dhaka experienced ankle to knee-deep water on the street (Fig. 1). Heavy traffic jams occurred because of stagnant water on the streets, and daily activities in parts of the city were almost paralyzed. At present, other major cities in Southeast Asia facing problems from regular urban flooding are Ho Chi Minh City, Hanoi, Vietnam, and Phnom Penh, Cambodia.

Examples of places outside South/Southeast Asia that suffer from urban flooding are Fukuoka and Tokyo, Japan (2); Playa del Gandia, Spain (3); and Houston, Texas, U.S.—where the storm “Allison” in June 2001 caused urban flood damage of the order of \$2 billion at the Texas Medical Center in the Harris Gully watershed (4).

THE IMPACT OF URBAN FLOODING ON SOCIETY

The costs of urban flooding are often very difficult to estimate precisely. Urban flood damage may be divided into three groups (5):

- direct damage—typically material damage caused by water or flowing water
- indirect damage—traffic disruptions, administrative and labor costs, production losses, spreading of diseases
- social consequences—negative long-term psychological effects, due to decrease of property values in frequently flooded areas and slowed down economic development.

One way to quantify the cost of urban flooding from structural damage is to collect information about damage costs from insurance companies, as has been done in Norway (5). This process is not likely to be applicable to developing countries, where few families in flood-prone areas have insurance.

During urban floods in developing countries, it is often the poorer people who suffer the most, as they often live in areas of cheap or free land, which are more exposed to natural disasters and have no or only primitive drainage and a drinking water infrastructure. Open drains of insufficient capacity in such areas are often flooded causing material damage and danger of epidemics. The water depth in some inundated city areas may typically be up to the order of 40 to 70 cm, which creates considerable traffic congestion, infrastructure problems, and huge economic loss in production, as well as significant damage to property and goods.

Diseases spread when sewage is mixed with surface flood water which, in turn, may reach and contaminate local drinking water sources. This unfortunate interaction between drinking water, groundwater, and floodwater causes severe health problems and calls for an integrated water management policy. Examples of diseases are diarrhea or leptospirosis, which are spread by bacteria in the urine of rats. In the September 2000 flood in northeast Thailand, 6,921 cases of leptospirosis were reported; 244 of these resulted in loss of human life (6). Last, but not least, parasites seem to thrive when urban flooding occurs

regularly. Moist soil provides a good environment for worm eggs to flourish, and flooded open drains spread eggs to new victims (1). Today, anthelmintics are available to kill parasites, but the parasites may gradually develop resistance to the drug and create new and more severe problems. The best way to manage parasite problems is to interrupt the life cycle of parasites, that is, to remove their natural environment by reducing the frequency and duration of flooding. For example, Moraes (7) found that reduced flooding decreased the prevalence of roundworm and hookworm by a factor of 2 and hookworm alone by a factor of 3.

WHAT CAN BE DONE TO UNDERSTAND AND REDUCE URBAN FLOODING?

The water-related infrastructure in cities consists of drinking water treatment facilities, water supply networks, sewer systems, and wastewater treatment plants. Many cities are old, and they have developed according to varying historical needs and visions. The sewer and drainage systems play the key role in efficient removal of storm water after rainstorms and prevention of urban flooding and its consequences. These systems, therefore, represent an important part of the urban infrastructure which was put in place through more or less continuous investments during the period of modern city development. This means that the layout and design of the infrastructure have gradually developed into rather complex systems. Inadequate planning and design as well as bad maintenance of these systems may cause severe economic and environmental damage in the cities, as many urban flooding cases prove. The problem is often aggravated if rainstorms coincide with high water levels in the sea or adjacent rivers during the monsoon season.

Many cities in the Western world manage local and minor flooding problems by using computer-based solutions. This involves building computer models of the drainage/sewer system, for instance, by using software such as MOUSE (8) and SWMM (9). Both of these software packages have been applied successfully to modeling major urban flooding events. The models are used to understand the often rather complex interaction between rainfall and flooding. Once the existing conditions have been analyzed and understood, alleviation schemes can be evaluated and the optimal scheme implemented.

At present, there are few studies on urban flooding that deal with both the conditions in the surcharged pipe network and extensive flooding on the catchment surface. However, it is feasible to model urban flooding based on the interaction between the pipe system and surface flooding, and this raises new possibilities for managing urban flooding problems. Examples of this kind of modeling are Bangkok, Thailand (10); Dhaka City, Bangladesh (11); Fukuoka and Tokyo, Japan (2); Harris Gully, Texas, U.S. (4) and Playa del Gandia, Spain (3).

To model urban flooding, a hydrodynamic model must be applied, and the flow and water levels in the major system (the street network and surface storage between the houses) and the minor system (the pipe network) must be described accurately. The major and minor systems

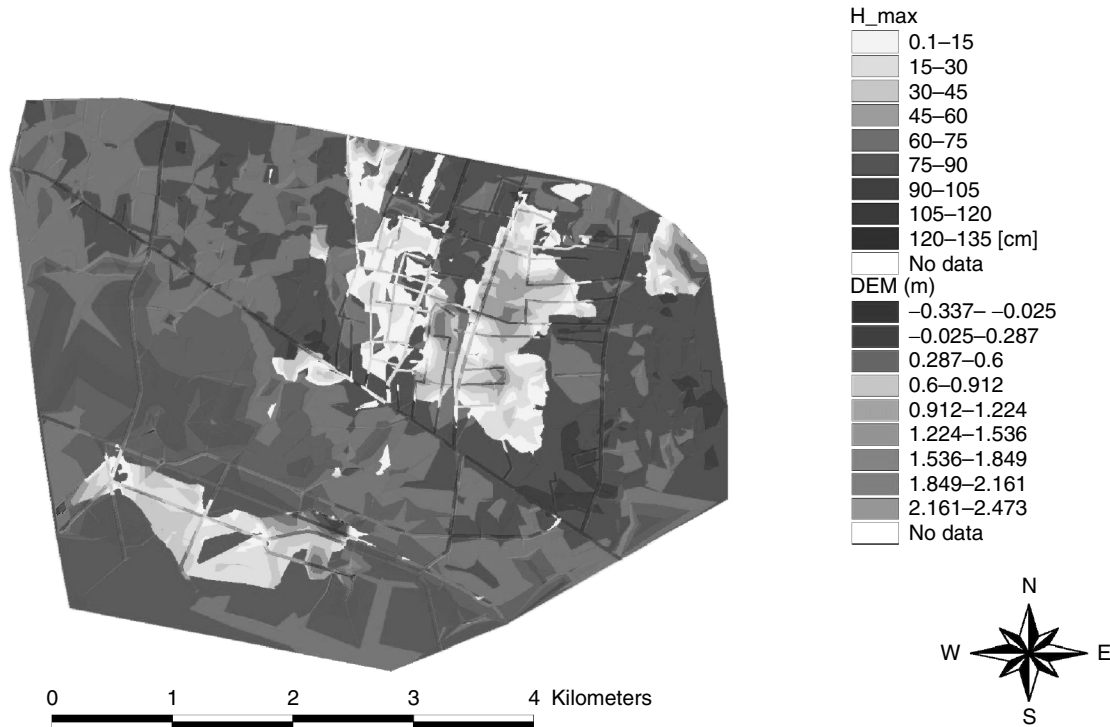


Figure 4. A flood inundation map for Bangkok (10).

route the rainfall runoff simultaneously in the pipes and on the streets. An exchange of water from the pipes to the streets (and vice versa) takes place, depending on the local hydraulic conditions in the catchment.

GIS plays a key role in data management for modeling urban flooding. A digital elevation model (DEM) is a powerful GIS tool for defining the catchment delineation and the storage of water on the surface. The DEM serves the dual purpose of input generation and result presentation of model simulations in terms of water depths in streets and between houses. An example of a flood inundation map for Bangkok (10) is shown in Fig. 4.

The handful of successful urban flood modeling applications carried out by computer models, which have traditionally been used to address local flooding problems in the industrialized part of the world, is very encouraging. The methodology is generic and can easily be transferred to other cities, if they have data available on the sewers and streets, as well as topographical maps.

FUTURE PERSPECTIVES AND CHALLENGES IN RELATION TO URBAN FLOODING

In 1997, the European Union (EU) passed a new and common set of directives, EN 752, for all EU member countries. This new standard calls for a new approach to the design of sewer systems in Europe. The old design criterion—design based on the return period of the rainfall—is replaced by a criterion based on the return interval of flooding. This new directive has boosted interest in understanding, research, and subsequent alleviation of urban flooding. Hence, it is foreseen that many urban flood applications will appear in the near future.

In the future, problems related to urban flooding will expand, as the cities in the developing countries are growing rapidly. So urban flooding must be seen in a broader perspective. An example is Dhaka City, which relies heavily (up to 97%) on groundwater for its water supply. During the last 25 years, the groundwater table has dropped by about 25 m (12). If this drop in the water table continues, it may generate problems for the city's water supply, and surface waters may need to be considered as additional resources. This is complicated by the growth of Dhaka City—areas that used to have permeable soils are being transformed into hard, impermeable surfaces. Such impermeable surfaces prevent replenishing groundwater storage and further aggravate groundwater problems. In turn, the runoff from the new impermeable surfaces generates additional surface runoff, which again increases the flooding in Dhaka City. Water supply pipes may be under low pressure during a period of flooding (e.g., due to power cuts), and polluted floodwater may enter the water supply network.

This poses an additional health risk to the population on top of the diseases spread by floodwater. In addition, there are considerable losses from the water supply network, so quite a large amount of drinking water is lost. This means that when dealing with urban flooding, one must remember that it involves not one problem, but a group of strongly interrelated problems, which have major impacts on the quality of life of people who live in flood-prone cities.

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FLOODWATER SPREADING

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INTRODUCTION

Providing usable water resources, especially in arid and semiarid regions, is going to be one of the major challenges of the twenty-first century. Therefore, alternatives in water resources development must be studied thoroughly as population growth increases the pressure on renewable and nonrenewable water resources. Most of the precipitation and surface runoff in areas subjected to water scarcity usually occur in seasons that do not coincide with periods of high water demand. On the other hand, growth in drinking and irrigation water demand has caused overuse of more reliable resources, such as groundwater, and has degraded their quality.

Flood spreading is a practical water and soil conservation technique that, if designed and maintained properly, can address one or a combination of water-related issues on local scales. Such issues may involve reduction of flood damage downstream by diverting a portion of the flood volume, controlling sheet and gully erosion over the land by maintaining vegetation cover, storing water in the soil profile or in underground aquifers for future use, improving soil fertility through sediment deposition, leaching saline soils, and irrigating cultivated and cropped lands where the rainy period and the growing period nearly coincide. The suitability and success of a flood spreading system depends on the project objective(s), nature of the soil, amount and temporal distribution of precipitation, land topography, and water demand.

Although flood spreading has been practiced widely with diverse objectives, the aim of this article is to present an overview of flood spreading schemes designed mainly for groundwater recharge and/or flood irrigation. "Water spreading" and "flood spreading" are used interchangeably in most literature, and we focus mainly on systems designed to operate on spate floodwater.

BACKGROUND

According to the former Soil Conservation Service (USDA, undated): "Water spreading is a specialized form of surface irrigation accomplished by diverting flood runoff from natural channels or watercourses and spreading the flow over relatively level areas. The diversion and spreading is controlled by a system of dams, structures, dikes or ditches, or a combination of these, designed to accommodate a calculated rate and volume of flow." Bennett (1) believes that water spreading has been limited largely to arid and semiarid areas used for range rehabilitation and crop production. He considers flood spreading a simpler and less expensive irrigation method than indirect irrigation with the aid of water stored in reservoirs.

Stoddart et al. (2) reported that spreading water schemes helped to increase forage production in the U.S. Great Plains and Southwest. The schemes consist of dams and dikes to divert and convey water to the land where it is absorbed. Heady (3) considers water spreading systems for spreading floodwaters to irrigate land, reduce and store sediment, or store water underground. For irrigation, a 1:20–30 ratio of spreader land to watershed area was suggested based on the work in Israel on the basis of 15% runoff, an average annual rainfall of 100 mm, and a 400-mm crop water demand. The rainfall threshold to generate runoff is increased for larger watersheds, thus, favoring small spreader systems. But the sediment per square area of watershed tends to increase for smaller watersheds. Heady (3) indicates that many such systems have failed due to inadequate design and the lack of maintenance.

Gupta et al. (1995) studied different water harvesting and conservation techniques. Their findings indicate that the benefit of water harvesting is high and shows a considerable increase in total biomass, tree height, root mass, and water use efficiency compared with the control plot.

HISTORY

The origin of flood spreading systems is not transparent in the literature. Bennett (1) states that floodwater use for crop production probably originated in arid regions where river overflow was used to water crops planted in adjacent lands. Writings and remains of irrigation dikes show that western South America, Mexico, and southwestern United States were some of the places where floodwater was used for farming.

Hudson (5) provides an overview of the traditional use of runoff in areas where rainfall amount does not meet crop demand. In parts of Arizona where the annual rainfall is 300 to 400 mm, Indian tribes used runoff from sandstone outcrops to water crops such as maize, squash, and melons planted at the base of hills (6). The Hopi tribe also applied flood farming in southwest North America and still continues this practice in smaller areas (7).

Kovda (8) reported the use of natural runoff in arid areas of the former Soviet Union to crop on flood terraces of large rivers. The technique is called "kair" farming. "Khaki" farming practiced in Turkmenistan allows runoff from mountain slopes to inundate the plains. After the plains have dried out, the land is cultivated and planted. Carr (9) reported similar evidence in Ethiopia. In a mainly pastoral community in Kenya where annual rainfall is less than 200 mm, small patches of sorghum are grown using a variety of runoff farming (10). Farming on valley bottoms is an ancient practice in Tunisia and the Negev desert (5), where annual rainfall is low, and in the level pans of one to three hectares formed in broad valleys of gentle slopes in Colorado, United States, where rainfall is 400 mm (11).

Runoff farming on level fields surrounded by earth banks has been carried out in the Khost plain, Paktia province in Afghanistan, where the runoff from stony hills is collected and led down to the fields. In two experimental small-scale projects in Kenya, water from a catchment area is diverted into the cultivated area, partitioned by earth bunds, into gently sloped terraces to prevent erosion (5). Stone spillways are built into the bunds to pass water to lower terraces. Construction of large-scale inundation spreading schemes, called "ahars," have been reported in semiarid regions of India, namely, in Bihar and Uttar Pradesh. Runoff during monsoon season is retained on very gentle slopes by low earth bunds such that the moisture stored can support a 5-month crop grown following the end of monsoon. Weirs are built to allow excess storm water to pass. An advantage of this system is regular leaching of saline soil in semiarid areas. Other forms of inundation farming are also reported in the Jaisalmer district of Rajasthan (12), where an earth bund is constructed across the valley plain to store water from surrounding hills. The catchment ratio is at least 15:1 where annual rainfall is 165 mm.

In India and Pakistan, a canal that has an elevated bed compared to the river, may be excavated in the river bank with a flatter slope than that of the river (5). The canal is filled by the flood overflow and runs dry during low flow periods. Maintenance of the canal entry is necessary for removal of deposited sediment. The water carried by the canal is then distributed over irrigation fields.

Implementation of flood farming practices, both in large- and small-scale experiments, have been also reported in some parts of Sudan, Morocco, Syria, Yemen, Australia, Jordan, Brazil, and Iran. The FAO (13) reports that the irrigated area under floodwater harvesting (spate irrigation) totals about 2 million hectares in North Africa and the Middle East.

SUITABILITY CRITERIA FOR FLOOD IRRIGATION

Bennett (1) numbers a few criteria for the suitability of land in flood irrigation projects. The criteria include rich fertile soil that has favorable permeability, gentle and smooth slopes, and sufficient rainfall. Annual rainfall of less than 8 inches (200 mm), or rainfall less than 4 to 5 inches during the growing season, may preclude installation of flood spreading systems. Range forage or supplemental feed may be produced by using flood spreading schemes where rainfall ranges from 8 to 14 inches (200 to about 350 mm).

The moisture stored in the soil must be sufficient to supply the crops until the next rainy season. Therefore, soils of high storage capacity, ideally medium to moderately fine texture of sufficient depth are preferable. Coarsely texture soil is not suitable because of its high drainage rate and low water holding capacity. Prinz (14) suggests a range of annual rainfall between 150 and 600 mm and a catchment to spreading area ratio between 100:1 and 10,000:1 and more for flood irrigation. Land slopes between 1 and 2 percent are suitable, and the water should not carry a high sediment load.

FLOOD SPREADING FOR ARTIFICIAL RECHARGE

Storing water underground has the benefits of no or slight evaporation loss, large storage capacity and no need to construct structures, and less vulnerability to drought and contamination. Besides augmenting groundwater storage, one of the primary objectives of artificial recharge may be to conserve and dispose of floodwaters. As a major source of water, storm runoff and floods overflowing the river banks or diverted by canals may be spread and stored underground for future use. Water (flood) spreading is perhaps the most widely used man-made system of artificial groundwater recharge. Water is spread over the ground, so it infiltrates into the soil and recharges the aquifer. The efficiency of the spreading system depends on several factors such as the spreading area, duration of recharge, soil hydraulic properties, volume of spread water, and topography.

Todd (15) classifies water spreading methods into stream, ditch and furrow, pit, basin, and flooding, out of which the last two methods are briefly mentioned here. Basin, the most favored and widely practiced method, is formed by dikes or levees or by excavation. Water is normally diverted from a stream and led to a single or series of basins, where water spills to lower basins. Large lands with gentle slopes are required for basin spreading. Sedimentation and sealing is likely to occur when turbid water is used, so the upper basins act as settling areas.

Periodic scraping, disking, or scarifying may be performed to restore recharge rates.

In the flooding method, suitable for a low 1–3% gradient topography, water (flood) is diverted from a stream to the spreading area. Erosion must be avoided as the shallow water flows over the area. The preferable texture of the soil is as coarse as possible to allow a high recharge rate. The cost of construction and maintenance is usually low for this method, but sediment, evaporation, and high floods are major problems. The former problem decreases the system efficiency, and the latter causes damage to canals, embankments, and spillways. The flooding method applied for groundwater recharge is generally similar to the flood spreading systems designed for irrigation.

FLOOD SPREADING SCHEMES

Many different flood spreading systems have been constructed in different countries. Flood spreading systems, mainly for irrigation, may be divided into two types: flow type and detention type (16). Free drainage and flow of water occurs in the first category thanks to a gentle land slope, whereas water is retained in the second type systems for infiltration. The flow type system is further divided into syrup-pan, spreader ditch, and dike and bleeder. The detention type is also classified into manual inlet control and automatic inlet control. The following describes different flow type systems, as stated by Jensen (16).

In the syrup-pan flow system, a single spreader ditch at the upper end of the field picks up the water from the main watercourse. Water spills over the sides of the spreader into the field divided into some sections by contour dikes. While flowing down the field, it partly infiltrates and partly runs off to the next section downstream. Contour dikes are broken at one end to allow water to proceed to the next section. At the end of the last section, there is a waste way or a channel leading excess water, if any, back to the main channel. Figure 1 presents a typical sketch of the system. In Fig. 2, a spillway along a contour dike is shown during operation.

In a spreader ditch flow system, a number of ditches divert water from a stream. The slope of the ditches decreases toward the lower end. Dikes and bleeder flow systems are a modified type of a syrup-pan system. Water flows through dikes to lower sections of the fields via tubes or weirs installed at intervals along the dikes. Emergency waterways connect each section to the main stream in case the tubes are blocked. Detention type systems are designed to divert and spread long duration flows with control over the depth of water.

DESIGN FACTORS

Several factors must be considered in the design of flood spreading systems. The meteorology and hydrology of the watershed, which produces the runoff for spreading, is of particular importance. Flow duration curves as well as flood values and the sediment load are to be

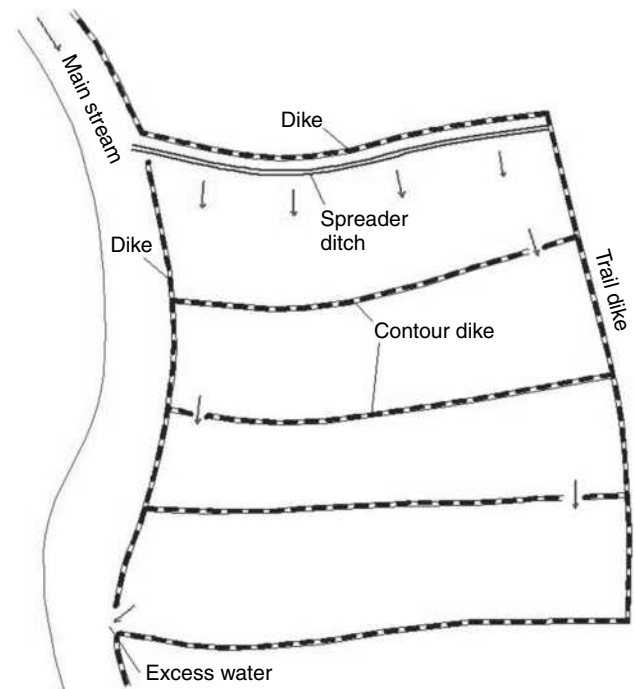


Figure 1. A typical syrup-pan flow system (modified from Reference 16).



Figure 2. A spillway along a contour dike during operation (Ghareh Baygon Flood Spreading Station, Fars Province, Iran).

studied. The major factors considered in the design of efficient spreading systems are diversion sites, slope of the spreading land, soil infiltration and water holding capacity, the area available for spreading, and economic considerations such as maintenance costs. Although a standard and widely practiced procedure for design of spreading systems does not exist, some irrigation and soil and water conservation books may include a chapter for designing them (16).

ADVANTAGES AND PROBLEMS

Flood spreading systems have the advantages of generally low construction and maintenance costs and adaptability to rural areas because of their simple technology. By

storing floodwater that would have been otherwise lost, local water demands can be partially met and flood damage downstream may be reduced. Other indirect benefits include environmental diversity, people migration control, and desert rehabilitation in arid areas.

The major problem of flood spreading projects is sedimentation. This decreases soil infiltration, which is a serious consequence in artificial recharge and flood irrigation systems. Efforts to maintain the efficiency of the system include growing vegetation, adding chemical soil conditioners, scarification, and leaving upper fields for siltation.

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MINIMUM ENVIRONMENTAL FLOW REGIMES

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INTRODUCTION

A growing demand exists for water by humans that not only reduced the amount of water available for future industrial and agricultural development but also has a profound effect on aquatic ecosystems and their dependent species (1).

Every inland, coastal, fresh, or brackish water ecosystem has specific water requirements for the maintenance of ecosystem structure, function, and the dependent species. For river-dependent aquatic ecosystems, normally referred to as environmental water requirements (EWR), environmental flow, or a new branch of hydrology, ecohydrology, (1) there is no universal definition of environmental flows. Flow is the governing influence in rivers, controlling many aspects of the physical environment and significantly affecting the biota of a riverine system. It is essential that adequate water be provided for the purpose of protecting the physical and ecological processes and features. Such flows are defined as EWR (2).

EWR are also defined as: “The water regimes needed to sustain the ecological values of aquatic ecosystems at a low level of risk” (3). Given that some systems are overallocated and unable to provide the full EWR, negotiation between stakeholders leads to the establishment of flows for the purpose of environmental protection. Such flows are defined as environmental water provisions, meaning: “The part of EWR that can be met” (3).

An environmental flow is the water regime provided within a river, wetland, or coastal zone to maintain ecosystems, and their benefits, where competing water uses exist and where flows are regulated. Environmental flows provide critical contributions to river health, economic development, and poverty alleviation. They ensure the continued availability of the many benefits that healthy river and groundwater systems bring to society (4).

Balancing the needs of the aquatic environment and other uses are becoming critical in many of the world's river basins as population and associated water demands increase (1).

In order to sustain the ability of freshwater-dependent ecosystems to support food production and biodiversity, environmental flows must be established scientifically, made legitimate, and maintained.

In recent years, a number of local and global assessments of water resources, current and future water use, water and food security projections, and water poverty analyses have been completed (1,2,5–10). Assessment of water availability, water use, and water stress at the global scale has also been the subject of increasingly intensive research over the course of the past

decades. However, the requirements of aquatic ecosystems for water have not been considered explicitly in such assessments (1).

OBJECTIVES OF ENVIRONMENTAL FLOW

The following environmental flow objectives apply specifically to the management of flow volumes in a river basin (5):

- a. Maintain appropriate minimum environmental flows (MEF) during low flow periods.
- b. Provide appropriate 'flushing flow' and 'high flow' regimes.
- c. Maintain, enhance, or restore species diversity, population structure, and community assemblages of aquatic biota and associated wildlife dependent on the instream environment.
- d. Ensure identified populations of significant biota are adequately protected through provision of appropriate environmental flows.
- e. Maintain and improve aquatic habitat, river structure, riparian vegetation, and water quality through provision of adequate environmental flows.
- f. Provide a flow regime that preserves and, where possible, reinstates important riffle and pool habitats.
- g. Minimize sedimentation and smothering of aquatic habitats through protection, maintenance, and restoration of indigenous riparian vegetation.

METHODS OF DETERMINING EWR

A range of environmental flow determination techniques exist and have been variously applied all around the world (4,11,12). Possible methodologies for calculating environmental flow requirements include:

Hydrological Methods

These methods are based on historical records and are simplest and the least data intense for estimating instream flows. They include the look up tables and desktop analysis.

Habitats Methods

They require some fieldwork to quantify the relationship between the selected variables describing the physical habitat behaviors and the river discharge. These methods are based on hydraulic simulations and habitat rating or modeling.

Holistic Methods

They are discussion-based approaches. The approaches offer a structured framework to the setting of instream flow requirements, based on expert opinion and stakeholder requirements. Holistic approaches make use of teams of experts and may involve participation of stakeholders, so that the procedure is holistic in terms of interested parties as well as scientific issues. Where methods

have the characteristic of being holistic, they clearly have the advantage of covering the whole hydrological-ecological-stakeholder system. The disadvantage is that it is expensive to collect the relevant data.

For a review of these methods, reading of Kinhill (13), Arthington and Pusey (14), Arthington and Zalucki (15), and Arthington et al. (16) articles are suggested. The determination of environmental flow requirements requires consideration of the entire flow regime. However, a key distinction in how environmental flow requirements are determined exists between low flows and high flows including floods.

When determining environmental flow requirements, it is important to understand the tolerance of stream biota to certain volumes, timing, and duration of flows, which can be achieved by comparing the current flow regime with the natural flow regime.

REQUIRED DATA

- a. Physical description of basin
- b. Water quality conditions in basin
- c. Description of key biota or environmental values (fauna and flora)
 - i. Fish
 - ii. Other vertebrates
 - iii. Aquatic macroinvertebrates
 - iv. Vegetation
- d. Water resource development and hydrology
 - i. Licensed water use
 - ii. System operation
 - iii. Stream flows

SOME RECOMMENDATIONS

Harby et al. (17) used a river system simulator analysis to optimize the environmental flow requirement of regulated river Maana in central southern Norway. The analysis recommended an environmental flow release of around 345 MLD (million liters per day). Zampatti and Lieshke (18) recommended 1.5 MLD as MEF for Diamond Creek Catchment, Australia. Zampatti and Close (6) recommended a MEF of 130, 8, and 9 MLD for Kiewa River, Running Creek, and Yackandandah Creek, Australia, respectively. Lieshke et al. (5) recommended maximum MEF of Plenty River system about 1.5 MLD. MDNR (19) recommended a MEF of 380 MLD for Potomac River system, Maryland. Smakhtin et al. (1) estimated EWRs needed to maintain a fair condition of freshwater ecosystems range globally from 20–50% of the mean annual river flow in a basin. These MEFs have been determined on the bases of the habitat and hydrologic-hydraulic methods. Arasteh (20) recommended a volume of 3400 million cubic meters per year as the annual minimum volume of EWR to survive Hamoon, Iran-Afghanistan trans-boundary wetlands on the base of a simple water balance method.

SUITABLE SITES

<http://www.g-mwater.com.au>

<http://www.cgiar.org/iwmi/assessment/index.htm>

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FORENSIC HYDROGEOLOGY

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Forensic hydrogeology is a branch of environmental forensics, a relatively new discipline involving the use of environmental science to support litigation.

Forensics is derived from the Latin root *forensis*, meaning “of a forum,” and is the study of formal debate. Accordingly, forensics involves arguing a position within any adversarial context. A legal context is not necessarily a requirement, but environmental forensics often support the interpretation or establishment of technical facts related to litigious matters. Generally, the product of a forensic hydrogeologic investigation is an opinion rendered in writing or orally via various legal vehicles such as affidavits, depositions, or trial testimony. The opinion may be direct or in rebuttal to an opinion proffered by an opposing expert. Due to the technical complexities of many legal matters pertaining to environmental issues, hydrogeologists often find themselves as members of interdisciplinary litigation support teams. These teams may comprise other experts in fields such as toxicology, chemistry, microbiology, engineering, industrial hygiene and safety, regulatory compliance, and industrial history and standards. Typical tasks for hydrogeologists include delineation of the source(s) and extent of contamination in subsurface soils and groundwater, interpretation of subsurface conditions controlling such flow and transport, analysis of groundwater flow and contaminant transport, determination of risk pathways toward potential receptors, and feasibility assessment of various remedial alternatives. The opinions rendered by the hydrogeology expert on these matters

may be corroborated or supplemented by the opinion of the other experts involved.

There are also forensic applications in hydrogeology that involve litigation, but which do not involve contamination or toxic tort matters. Examples include settlement of damages between parties involving water supply issues, such as well interference or interruption of baseflow; flooding, such as watershed or hydrographic modifications resulting in increased runoff; or simply well drilling and design, involving poor yields or failures.

FORENSIC ISSUES IN HYDROGEOLOGY

There are a variety of legal issues whose resolution have required the support of forensic hydrogeologists. Recent cases have involved supporting determinations of allocation for environmental remediation costs among multiple responsible parties, circumstances "triggering" insurance policy coverage issues, environmental liabilities accompanying property transfers, and personal damages or tort claims from alleged exposure to contamination released from a site. Other claims have arisen over performance of contractors working on environmental cleanup or who were engaged in the construction or operations and maintenance of facilities that failed and allegedly resulted in contaminant releases.

Determination of fair and defensible cost allocation among parties potentially responsible for groundwater contamination is a complex undertaking. Various approaches involving allocation on the basis of area-, volume-, and mass-of-contamination, as well as weightings based on such factors, have been proposed (1,2). Additional approaches based on toxicity weighting have also been proposed for sites where multiple-groundwater contaminants are commingled. Such commingled plumes may also give rise to issues with regard to selecting alternative remedial technologies.

Depending on the period and type of coverage, exclusions, and various interpretations of what constitutes an occurrence, insurance claims involving liability for environmental damage may generate issues requiring input from hydrogeologists. For example, there may be matters related to the timing of releases relative to the timing of the policy coverage or to the enactment of certain "pollution exclusion" clauses within the policy. In such instances, the hydrogeologist may be able to opine on the likelihood that contamination occurred during the policy period and whether the contamination is persistent, requiring present-day remedial efforts.

Property transfer issues may be similar to the allocation and timing considerations considered earlier. Such matters require careful research of historical activities performed on the property. Hydrogeologists might use record searches, interviews, and review of historical photographic archives to reconstruct chronologies and thereby form opinions on the likelihood of certain operations as causes for contamination.

Tort claims are often the most complex issues and require extensive examination of risk factors from the source of the contamination, the transport pathway, and the point of exposure. Tort damages may not be limited

to human exposure but may also extend to resource damage. In such matters, hydrogeologists might be able to help clarify the timing and fate of contamination from the source to the site of exposure using analytical or numerical models based on conceptualizations of the site hydrogeology.

TOOLS AND APPROACHES

In theory, the approach of hydrogeologists serving as experts in litigation is not fundamentally different from their serving in any other professional capacity. In other words, the tools of the trade are the traditional ones, and the approach involves traditional scientific methods of investigation. There are, however, issues that arise in contemporary forensic matters that are more interdisciplinary. In matters involving contaminant liabilities, environmental forensic specialists have been referred to as "industrial paleontologists," signifying their proficiency in unraveling the history of events leading to present-day conditions at a contamination site (3). Having been trained extensively in geology, hydrogeologists are particularly well suited for such "after-the-fact sleuthing."

As in any traditional investigation, forensic study is staged. The first stage is the acquisition and organization of pertinent data and information. This may involve simply compiling records, photoarchives, and searching databases, or it may include actual field data acquisition such as drilling, sampling, and analysis. At this stage, the hydrogeologist has various traditional and specialized tools available. For example, there are methods traditionally employed such as water level measurements and water and soil sample collection and analysis. Contaminant investigations may also incorporate isotopic analysis or chemical fingerprinting to assess timing or source issues to the extent materials released have a unique, detectable chemical signature. The basis for chemical fingerprinting might be weathering characteristics and composition of petroleum hydrocarbons, proprietary additives, alkyl-lead compounds, oxygenates, or isotopic ratios. Hydrogeologists might also consider the changing state of industrial technology. For example, much information has been gathered on the historical use of organic solvents (4). Some solvents were commercially available and in use at different times, as indicated in Table 1. Certain solvents were more prevalent in industrial use than others at different times. Other chemicals were banned for certain usages by regulatory agencies. A hydrogeologist who is knowledgeable about historical chemical usage/availability and can reconcile such information with chemical use and disposal history of the site can present informed opinions on the timing of releases. Using chemical associations in this manner and context is analogous to using index fossils in geochronology.

The next stage involves assessing and analyzing the data per the study objectives. This analysis is based on both the hydrogeologist's conceptualization or alternative conceptualizations of the hydrogeologic system and the impacts of site operations on the soil and groundwater. The hydrogeologist assembles information gathered to determine such things as the occurrence, flow direction,

Table 1. Year of First Commercial Availability of Selected Organic Chemicals^a

Chemical	Year of First Commercial Availability
Carbon tetrachloride	1907
Trichloroethylene (TCE)	1908
1,2-Dichloroethane (DCA)	1922
Tetrachloroethene (PCE)	1925
DDT	1942
Chlordane	1947
Toxaphene	1947
Aldrin	1948
Dibromochloropropane (DBCP)	1955
Bromacil	1955
<i>Gasoline additives</i>	
Tetraethyl lead	1923
Methyl cyclopentadienyl manganese tricarbonyl (MMT)	1957
Tetramethyl lead	1966
Methyl tert-butyl ether	1980s

^aReference 3.

and use of groundwater; interaction with surface water; geologic cross sections; distribution of contaminants in the soil and groundwater; and other factors related to the overall hydrogeologic conditions at and in the vicinity of the site. Current and past operational information can then be used to interpret the contaminant history of the site. The approach for this step can vary widely according to the skills and experience of the expert as well as the history of site operations and the type and availability of site records. In some cases, the hydrogeologist may want to apply various models to simulate or project groundwater flow and solute transport conditions. Other types of modeling that are less familiar to traditional hydrogeology, but which can support interpretations on timing of release, might include various geochemical models, degradation models of various fuels and components, and corrosion models (for tanks and pipeline leaks).

ETHICS AND STANDARDS

In the process of performing these services, extreme care must be taken to organize and document the information used and generated, considering the likely degree of scrutiny by both technical and legal adversaries. In general, this requires strict adherence to professional ethical standards and methods of analysis, systematic maintenance of records, traceability, and disclosure.

With specific regard to the admissibility of expert opinion in the courtroom, some courts use criteria or standards to determine the acceptability of expert testimony. One such precedent was set in the opinion of *Daubert v. Merrell Dow Pharmaceuticals, Inc.*, which held that judges must function as gatekeepers in assuring that expert opinion proffered in the courtroom is valid and reliable (5). To this goal, the court suggested criteria based on the following questions:

1. Is the opinion testable and has it been tested?
2. Is the error rate associated with the technique or opinion acceptable?
3. Has the basis for the opinion survived peer review and has it been published?
4. Is it generally accepted among scientists in the pertinent field?

Other courts have simply applied a “general acceptance” criterion, which originated in the 1923 case of *Frye v. United States*, which dealt with admissibility of a polygraph-type examination. This court held that the scientific opinion must have achieved “general acceptance” in its particular field or context.

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FORESTS AND WETLANDS

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Forests and wetlands are landscape features that play integral roles in the hydrological cycle. Within the landscape, forests and wetlands are inextricably linked. Wetlands may exist within forest ecosystems, or wetland ecosystems can be characterized as “forested.” This article will outline the characteristics of both forests and wetlands and some implications for water resource management.

FORESTS

Forests cover an estimated area of 4.17×10^9 hectares, which is roughly 32% of Earth’s surface (1). A forest can be defined as “a biotic community predominated by trees and woody vegetation that covers a large area” (1), whereas others define forests by its vegetative character regardless of area. Forests serve social, economic, and ecological functions, which impact water resources in

myriad ways. A salient economic use of forested land throughout the world is as a source of fuel and timber. The various hydrological functions that forests perform have substantial economic benefits as well, which ecologists and economists have tried to quantify in recent decades. These economic factors may be a driving force in forest and water resource management decisions.

DISTRIBUTION OF FORESTS

Forest types vary along temperature and moisture gradients. Forests generally occur in regions where the annual precipitation exceeds 38 cm, the frost-free period is at least 14 to 16 weeks in duration, and solar radiation is greater than 27 W/m^{-2} (1). Regional conditions, such as ocean currents and air mass circulation patterns, may also determine the range of forest types in a particular location.

Forested landscapes impact global hydrologic systems by vapor and gas exchange, and conversely, these hydrologic regimes impact the distribution and composition of forests. For instance, overharvesting of timber in semiarid regions may result in lower precipitation, thus further constraining the growth of vegetation (2). On a local scale, the composition of a forest will impact processes such as throughfall, residence time, chemical enrichment, litter characteristics, and soil characteristics (3).

FOREST STRUCTURE AND THE HYDROLOGIC CYCLE

Interception

Precipitation, or the deposition of liquid or solid water onto the Earth's surface, can fall directly onto the soil surface, be intercepted by vegetation or other surfaces, or flow to the ground via trunks and stems (4). The forest canopy plays a significant role in precipitation interception. It has been found that 10–20% of annual precipitation is removed by canopy interception, depending on climatic and vegetation characteristics (1).

Litter on the forest floor may also intercept and store water temporarily before it infiltrates into the soil. The leaching that occurs as water travels through the litter layer can have a substantial impact on the biogeochemistry of the site. The enrichment of precipitation intercepted by litter will vary among forest types because of differences in litter composition. On average, the litter layer in deciduous forests is thinner than in coniferous forests (3), therefore constituents (i.e., nitrogen, calcium) have been found to leach out in higher quantities at sites dominated by deciduous species as compared with conifer-dominated locations (3).

Evapotranspiration

A portion of the intercepted water may evaporate and return to the atmosphere. The rate of evaporation is determined by the amount of water present and the amount of solar energy available (5). Transpiration can also result in the loss of water vapor into the atmosphere. Transpiration is the process whereby water is released from plants' stomata on leaves and lenticels on stems.

Transpiration rates can be a considerable output of water in a watershed depending on the type of vegetation and the age of the stand. On average, 95% of the water taken up by a plant's roots will be transpired, whereas the remaining 5% will be converted into biomass via photosynthesis (5). Chang (1) states "a mature forest can transpire as much as 1000 tons of water to produce 1 ton of wood," while noting that this figure can be even higher in some species. As a result of the difficulty of empirically differentiating the water lost by evaporation and that which is lost by transpiration, these two processes are discussed as single process called evapotranspiration.

Throughfall and Stemflow

Precipitation is also routed toward the forest floor by the processes of throughfall or stemflow. Throughfall is defined as the portion of the water that falls without being intercepted or that which was stored in the canopy that drops from the canopy to the forest floor (5). Stemflow is the water that travels along the stems of plants. Throughfall and stemflow are important processes because they influence the distribution of and quality of the water that reaches the soil surface, and thereby they have a "[significant] impact [on] forest biogeochemical cycles" (6). Leaching of nutrients from above ground biomass in deciduous hardwood forests can account for up to 20% of the annual nitrogen and potassium fluxes (6).

Despite its small volumetric contribution to the forest ecosystem, stemflow, in particular, has been found to have a significant impact on runoff generation, soil moisture patterns, and soil water chemistry (6). The rate and magnitude of stemflow has been found to be correlated with vegetation type and structure, storm event characteristics, and seasonality. In general, stemflow yields increase with the magnitude of a storm event (6). The chemical inputs from stemflow also vary based on seasonal variables. Concentrations of target constituents were found to be greater during the winter months than during the summer. Changes in canopy density, greater residence time in the bark, and the lower bark pH during the winter are some of the seasonal factors (6).

Infiltration and Runoff Mechanisms

The forest type, and its subsequent root system, plays an important part in soil genesis and structure, as well as the soil's infiltration capacity. In forest ecosystems, 40–70% of the total biomass is found below the soil surface. Therefore, root systems introduce a significant amount of organic material into the system, which facilitates the formation of aggregates, the organization of which determines soil structure. The roots also remove soil moisture from the adjacent soil, which serves to increase soil aggregation (7).

In addition to the organic matter that is contributed by the forest root system, roots also move soil and form macropores that play an important role in infiltration and percolation. Woody plants can send their roots more than 5 m deep depending on the substrate (7). These roots often follow existing channels and voids, but as they grow, they expand these spaces. When roots die, these voids are left empty and can influence the hydraulic conductivity of the soil under certain conditions.

The type and structure of forests also affects the process of infiltration of water into the soil (1). The infiltration capacity is the "maximum rate at which precipitation can infiltrate into the soil" (5). In most forest systems, the infiltration capacity is generally high and infiltration excess overland flow does not generally occur and cannot be considered a significant runoff mechanism (5). That being said, surface conditions such as vegetation cover, roughness, crusting, soil temperature, and slope can determine the quantity of overland flow and its velocity (1).

The hydraulic conductivity is the ability of a soil to carry water or another liquid (5). The conductivity of a soil depends both on the properties of the soil as well as the properties of the liquid. The soil structure and the system of macropores will impact the conductivity. The water that firsts enters the soil after a storm event will follow preferential pathways. Typically, water infiltrating at the soil surface will follow the path of least resistance filling large pores, cracks, and animal burrows before the rest of the soil has been saturated. Forests, like was mentioned above, have a greater number of macropores because of root propagation and the decomposition of dead roots.

In addition, a healthy root system can reduce the amount of soil lost in overland flow events. Thus, soil stabilization and retention can prevent erosion and retard surface water pollution.

Management Implications

As a result of the integral relationship between forests and the hydrological cycle, the extent and quality of forested areas will have an impact on the water balance of the watershed. On average, peak flows of streams are generally lower in watersheds that have a high percentage of forested area (1). Activities that reduce forest cover, such as residential and industrial development, clearing for agricultural operations, and logging, may change the hydrological balance within a given watershed. According to Huber and Iroumé (8), intensive forest operations can have implications for water yield and quality. Removal of forest vegetation can impact soil moisture, infiltration capacity, and total water yield. A decrease in interception capacity, by removing or altering the forest canopy, can result in greater soil moisture.

For example, reduction of forest canopy area can cause an increase in snow accumulation on the ground. As the snow melts, an overall increase in water yield will occur because the storage capacity of the watershed has decreased. Also, an increase of impervious surfaces can decrease infiltration, contributing to a greater quantity of infiltration excess overland flow. Therefore, a greater quantity of water is released into streams.

WETLANDS

Wetlands are ecosystems inundated by water during the growing season, which produces soils dominated by anaerobic processes and in which the biota exhibits flood-tolerant adaptations (9). Wetlands cover approximately 4–5% of the Earth's surface. They exist in across a broad spectrum of climates from boreal peatlands to tropical

mangrove forests (9). Although they make up a relatively small percentage of the total land area, wetlands can have disproportional affects on hydrological and biogeochemical cycles in local, regional, and continental contexts.

FORESTED WETLANDS

Forested wetlands are the dominant wetland type in the continental United States (10). These wetlands are of particular interest because they lie at the interface of upland and aquatic systems; these wetlands possess characteristics of both upland and aquatic systems while having distinctive properties found only in wetlands. This unique landscape position makes forested wetlands important to surface and groundwater quality as well as biodiversity. The forested wetlands of most interest to this discussion are swamps and riparian wetlands.

Swamps are wetlands that are dominated by trees rooted in hydric, mineral soils (9,11). Swamps have standing water through most or all of the growing season (11). Freshwater swamps, or deepwater swamps, have a broad geographic distribution ranging from the bald cypress-tupelo (*Taxodium distichum*-*Nyssa aquatica*) swamps that are found from the Midwest to the southern United States to the Atlantic white cedar (*Chamaecyparis thyoides*) swamps and red maple (*Acer rubrum*) swamps of the northeastern United States. The hydrogeology and landscape position of swamps will, in large part, determine the hydrologic and ecological functions that the wetland performs as well as the magnitude of these functions (11).

The riparian zone is the corridor of land on the banks of streams, rivers, and lakes. Riparian wetlands are found within the riparian zone, in areas that receive seasonal inputs of water through flooding of the adjacent body of water, but that do not have standing water throughout the growing season. Although riparian wetlands are not necessarily forested (i.e., wet meadows), they often are. Willows (*Salix* spp.), red alder (*Alnus rubra*), and green ash (*Fraxinus pennsylvanica*) are common bottomland species found in riparian wetlands. Riparian vegetation, particularly riparian forests, has a significant affect on stream geomorphology and ecology. It has been found that stream water temperatures, for example, are correlated with riparian soil temperatures. Riparian vegetation also provides allochthonous inputs that are used by aquatic biota as a source of carbon.

ROLE OF WETLANDS IN WATERSHED HYDROLOGY

Wetlands, especially forested wetlands, perform similar hydrologic functions to those performed by upland forests. Interception, evapotranspiration, throughfall, and stemflow are all processes that occur in wetlands. However, unlike upland forests, wetlands are more directly impacted by the hydrologic environment. The character of wetlands is directly affected by specific effects of hydrology (11). Hydrology leads to specialized vegetative communities; also, flow properties can determine productivity, and nutrient cycling is impacted by the hydrological characteristics of a wetland (11). Wetlands

also provide important hydrological functions, including hydrological storage, biogeochemical cycling, and pollutant retention (9).

Hydrological storage includes flood attenuation, groundwater recharge, and in some cases, groundwater discharge. Wetlands attenuate flood waters by intercepting storm runoff and flood waters, storing the water for some period of time, and then releasing the water at a slower rate than if it had not been intercepted. Thus, wetlands reduce the peak flows and spread them out over time (11). The abatement of floods by wetlands has both economic and ecological implications. The reduction of peak flows also has water quality benefits. Wetlands reduce the velocity of the peak flows, which allows sediments and other pollutants to settle out from the water column. Flood reduction can also decrease scour of stream bed and banks, reducing the amount of material eroded from stream channels.

Wetlands contribution to groundwater is largely dependent on the geomorphology of the wetland. Groundwater dynamics can play an important role in some wetlands, but be negligible in others. Groundwater recharge is common in depression wetlands such as prairie potholes in the mid-western United States. These wetlands fill with water during the spring, and as the growing season progresses, water from the wetland infiltrates the soil and percolates into the aquifer. In other wetlands, such as fens, groundwater is discharged into the wetland and is the dominate source of water for the wetland. In other wetlands, neither significant groundwater recharge nor discharge occurs (11).

Biogeochemical cycling is another crucial function of wetlands. Wetlands specialized role in biogeochemical cycling is largely because of the alternative aerobic and anaerobic soil conditions throughout the year (11). When the wetland soils are saturated, they become anaerobic and a series of reduction reactions occur. The reduction of compounds can alter their bioavailability and prevent their introduction into surface and groundwaters. When the soil moisture decreases and aerobic activity again commences, these constituents can become available again. A large body of research has examined the removal of nitrates (NO_3^-) from subsurface water and phosphorous (P) from overland flow in riparian systems (12–15). The efficiency of these wetlands to remove nutrients and sediments is largely because of vegetation characteristics, preferential flow paths, and seasonality of inputs.

Wetlands are sometimes referred to as the “kidneys of the landscape” because they can remove nutrients, particularly nitrogen and phosphorus, pesticides, bacteria, heavy metals, and other potential pollutants from the surface or groundwater that they intercept. Similarly, researchers often conceptualize the landscape as a series of sources and sinks. Within this framework, wetlands are often thought of as sinks for pollutants. In reality, the relationship between the wetland and its watershed is often much more complex. At a certain point, wetlands may become saturated with a certain constituent and become a source rather than a sink (16). For example, when a high sediment yield exists in a watershed, wetlands may become a source of sediment. In large basins that have high

upland yield, wetlands can actually account for 52–91% of the sediment discharged into streams. The role of wetlands as sources of sediment may be more variable in smaller watersheds, depending on the geomorphology and land use in the watershed (17).

The benefits of having wetlands throughout the watershed are now well known. Yet, anthropogenic disturbances as well as natural variation have contributed to the loss of wetlands throughout the world. It is estimated that, in the United States, the loss of 50% of the wetlands has occurred since European settlement (11). Wetlands are typically drained so that the land can be used for other uses, such as agricultural or residential development. The loss of wetlands from the watershed can translate to increased stream discharge, increased sediment yields, and eutrophication of surface waters.

Another anthropogenic disturbance that impacts riverine swamps and riparian wetlands is the reduction of flow in the stream channel or channel alteration. These disturbances impact the seasonal inputs of water from flooding and may cause changes in the composition and distribution of plant species. These changes may also impact the rate and magnitude of biogeochemical as well as hydrologic functions.

CONCLUSION

Although it may be helpful to look at forests and wetlands as two distinct ecosystems, it is important to understand the interconnections between them. Forest management regimes may determine the types of constituents as well as the amount of water that travels into bottomland areas by overland flow as well as subsurface flow paths. Subsequently, the condition of wetlands can have an effect on the storage and cycling of nutrients, sediments, and organic matter. The consideration of both of these ecosystems when making management decisions will enable managers to accentuate the ecosystem functions of one ecosystem without compromising the health of the other.

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ROCK GLACIER

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Active rock glaciers are the geomorphological expression of creeping mountain permafrost (Fig. 1). They consist of a debris–ice mixture that mechanically deforms and slowly moves downslope or downvalley and develop therefore a tongue-shaped or lobate landform with a front scarp and characteristic surface structures, such as furrows and ridges, as rheological expressions. Unlike “true” glaciers, rock glaciers do not present surface ice or exposed ice in general but contain only supersaturated ground ice.

Rock glaciers that do not move any longer but still contain ice are called inactive, and the sediment body that remains after a rock glacier has melted is called a relict (or fossil) rock glacier. Rock glaciers may become inactive for climatic or dynamic reasons. In the former case, the seasonally unfrozen block mantle has grown as a consequence of permafrost degradation. A dynamically inactive rock glacier, in contrast, may develop even in areas of continuous permafrost if it enters into flat terrain or if its thickness falls below a critical value. The front scarp of active rock glaciers generally reaches inclinations between 35° and up to 45°, but inactive ones show front

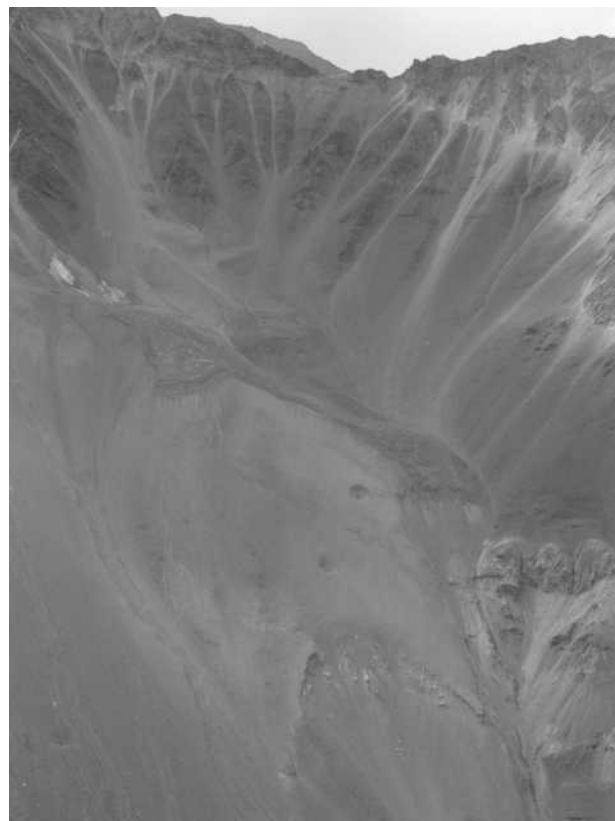


Figure 1. An active talus rock glacier in the Andes of Central Chile (Estero de las Yeguas Muertas, 33.5°S). (The crater-like depressions are produced by subsrosion of gypsum).

slopes at or below the angle of repose, and the upper part of inactive front slopes is gentler than for activity. Relict rock glaciers can be recognized from their subdued appearance and irregular surface structure. Further classification criteria are presented in Table 1.

In the 19th century, several authors already described rock glaciers, but only with the work of Wahrhaftig and Cox (1959) did modern and more comprehensive rock glacier research begin. However, until today, some confusion persists, mainly of the terminological kind concerning the precise differentiation of rock glaciers and glaciers (Table 1).

The development of rock glaciers is a consequence of the enrichment of debris with percolating snow meltwater, which freezes under permafrost conditions and forms interstitial and segregation ice. Therefore, rock glaciers are of periglacial origin. They may form out of talus accumulations (talus rock glaciers) or morainic debris (debris rock glaciers).

Rock glaciers contain between 40% and 60% of ice by volume and therefore constitute unsuspecting stores of water. In the Swiss Alps, rock glaciers contain only 0.03 km³ of water equivalent per 1000 km² of mountain area, compared with 2.5 km³ per 1000 km² stored within glaciers. In semiarid mountain areas, such as the Andes at 33°S, in contrast, the amount of water stored within rock glaciers is by one order of magnitude higher, while at the same time glacier sizes are smaller.

Table 1. Rock Glacier Taxonomy as Suggested by Barsch (1987)

Type	Talus rock glacier (material: talus) Debris rock glacier (morainic debris) Special rock glacier (other material)
Location	For talus rock glaciers: –Valley head, including cirques –Valley side walls –Foothlope below cliff For debris rock glaciers: –Glacier terminus –Side of glacier
Connection to source area	Direct Not direct
Surface relief	Very well developed Subdued to well developed No furrows and ridges
Surface grain sizes	Earthy/blocky
Form	Singular/complex
Complexity	Multipart/multilobe/multiunit/ multiroot
Shape	Tongue-shaped/lobate/transitional
Size (area)	Small ($<10^4$ m ²), medium, huge ($>10^5$ m ²)
Activity	Active/inactive/relict

Therefore, rock glaciers are important water resources in high mountain areas of continental semiarid and arid climates. In spite of this, little is known about their quantitative importance for river discharge.

The lower limit of rock glacier distribution is associated with the lower limit of discontinuous mountain permafrost, which is often attributed to the -1°C to -2°C isotherm of mean annual air temperature. Therefore, rock glaciers are considered good permafrost and climate indicators on a regional scale.

Concerning rock glacier movement, horizontal displacement rates range between a few centimeters and 1–2 m per year. Velocities are highest in the central lower part of the rock glacier and decrease laterally and toward the rock glacier's rooting zone. Furthermore, rock glaciers seem to move faster in summer than in winter.

The age of active rock glaciers may generally range between several thousand years and up to 10,000 years. Although several attempts have been made to obtain more precise data, absolute age determination has turned out to be extremely difficult. As climatically inactive rock glaciers are often situated at the lower limit of the distribution of active ones and up to 200 m below this limit, their inactivity probably developed since the Little Ice Age.

Recent rock glacier investigations focus increasingly on their physical properties. Geophysical methods, such as geoelectric soundings and refraction seismics, are being used in order to differentiate between permafrost and unfrozen ground along cross sections through the rock glacier. Furthermore, boreholes have been established in order to directly observe the internal structure of rock glaciers and monitor the change of deformation rates with

depth. As the coarse boulder mantle of rock glaciers may strongly influence their energy balance, this issue is also being addressed within the more general framework of ground thermal regime in permafrost areas. At a regional scale, statistical methods are being applied in order to determine morphological and climatic controls on rock glacier formation.

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GREAT LAKES

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GEOLOGIC ORIGINS

Lakes Superior, Michigan, Huron, Erie, and Ontario comprise the five interior freshwater lakes that extend across a total area of 244,000 km² (0.16% of world land area), occupy parts of six states and two provinces in two nations, and contain nearly 20% of the world's freshwater supply (see Fig. 1). Glaciers, although frozen, are the only contiguous surface freshwater volume that contains more freshwater; they were the geologic forces that carved these lakes and the surrounding drainage area from the Precambrian rocks and overlying marine deposits from earlier inland seas that are now rocks classified by their Cambrian, Ordovician, Silurian, Devonian, Mississippian, and Pennsylvanian periods. The current Pleistocene Epoch of glacier activity, whose vertical heights were 2 km, compressed and scoured the area during numerous advances and retreats. During each icesheet retreat, the ice-excavated lake basins were filled by meltwater, and the absence of ice overload burden triggered areas of earth uplift and the creation of drainage outlets. Prior to the St. Lawrence drainage that followed the most recent

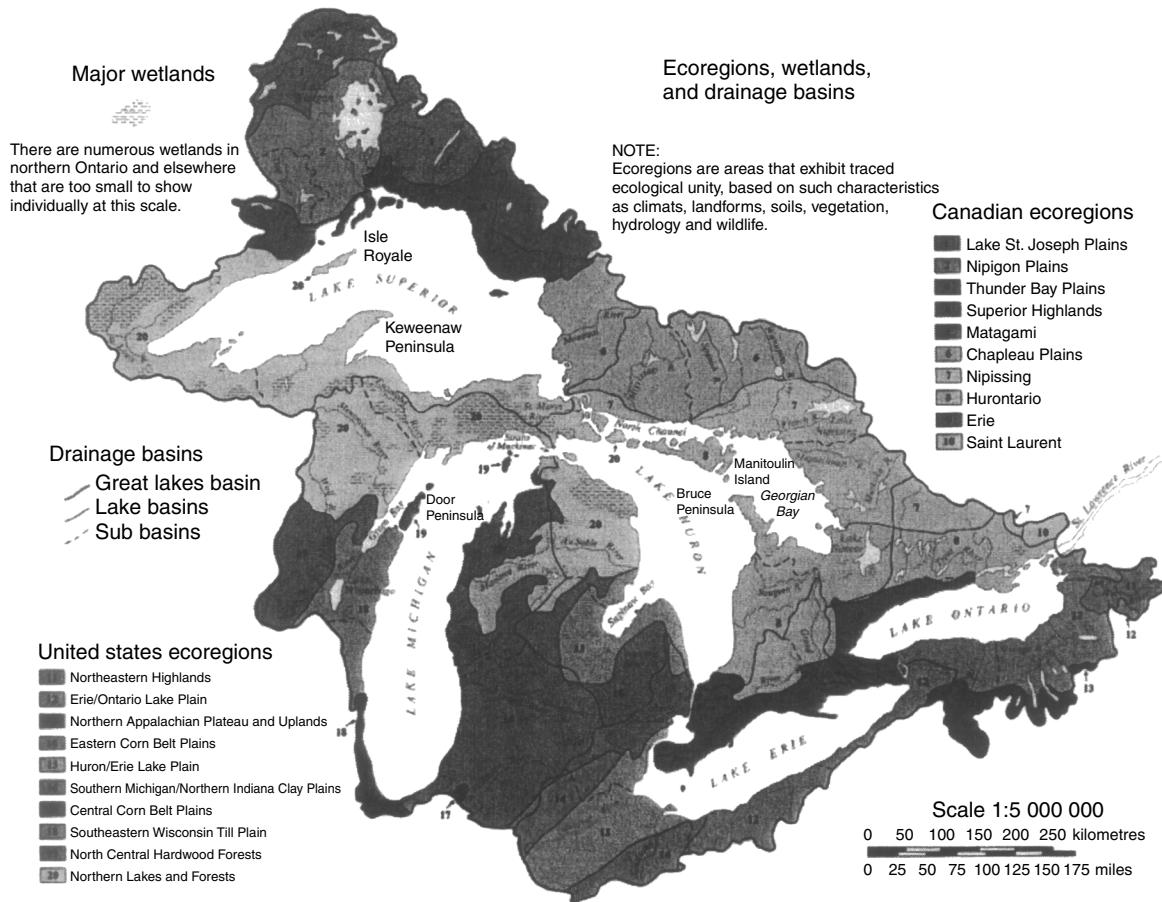


Figure 1. Great Lakes ecoregions, indicating unique features of the basin formed by geological forces, such as low-lying wetland areas.

Wisconsin icesheet retreat, lake drainage had spilled through Illinois River Valley toward the Mississippi, the Hudson River Valley, the Trent River, and the Ottawa River Valley. Uplift continues today, although more slowly.

Deposits from the glacial scour and retreat are largely poorly sorted mixtures of boulders, gravel, sand, silt, and clay, referred to as glacial drift and till. Exposed consolidated rocks of limestone, shale, sandstone, halite, and gypsum formed during the preglacial period of inland seas from sedimentary deposits are found in many river and glacial carved valleys; outcrops of igneous and other

morph rocks are also revealed. Landforms include sand dunes; flat till plains; linear moraines; ramp shaped drumlins; and meltwater deposited eskers, kames, and outwash. Relatively low relief defines the drainage basin, but the lakes feature remarkable depth gradients, widths, and lengths, which are recorded in Table 1. It is apparent that extremes exist among the lakes; Erie and Superior have nearly a 350 m difference in maximum depth. The lakes are stunning, but the wetlands occupying the low gradient lands, as well as the isolated gorges and ravines carved by active rivers, also provide both ecological and geologic complexity.

Table 1. Physical Features of System

Category	Superior	Michigan	Huron	Erie	Ontario
Elevation at low level, m	183	176	176	173	74
Length, km	563	494	332	388	311
Breadth, km	257	190	245	92	85
Average depth, m	147	85	59	19	86
Maximum depth, m	406	282	229	64	244
Volume, km ³	12,100	4,920	3,540	484	1,640
Water area, km ²	82,100	57,800	59,600	25,700	18,960
Land drainage area, km ²	127,700	118,000	134,100	78,000	64,030
Total basin area, km ²	209,800	175,800	193,700	103,700	82,990
Shoreline length, km	4,385	2,633	6,157	1,402	1,146

HYDROLOGIC FLOWS AND CLIMATE

Great Lake flows are from Lake Superior and Michigan into Huron, Huron into Erie, and Erie into Ontario, which flows into the St. Lawrence Seaway through Canada and into the Atlantic. Numerous smaller inland lakes, rivers, and wetlands drain into the Great Lakes system, including some famous waterbodies such as the Finger Lakes, Lake Nipigon, Lake St. Clair, the Niagara River, and the Montezuma Wetlands of New York. Alterations to flow have occurred, such as diversions of Long Lac and Lake Ogoki to capture water from the Hudson Bay watershed, dams to drown the Lachine Rapids, and reversing the Chicago River to keep sewage from mixing with drinking water. Interesting dynamics of flow within the Great Lakes system include fall and spring turnover that breaks the Lake's thermocline strata and mixes hypolimnion and epilimnion waters, seiche, or sloshing of lake water height from shore to shore, an oscillation due to the large fetch of lake surface and strong winds piling water against the coast, and devastating surface waves caused by gale force winds. The Welland Canal, a unique man-made feature of the system, forms a navigable channel between Lakes Erie and Ontario, which are vertically separated by nearly 100 m, a drop that is dramatically illustrated by the Niagara Falls.

Surface water resources are enormous in volume and area; the precipitation, evaporation, and runoff components of the hydrologic cycle maintain this system that has a large flux rate, as shown in Table 2. Runoff into the lakes is smallest for Erie and Ontario, almost half that for Superior and Huron, but is augmented by the inflows from those same upper Great Lakes. The retention times follow a logarithmic decrease of 100-year intervals from the headwater Lakes of Superior and Michigan, which have large volumes and small inflows, to the yearly intervals for Erie and Ontario, which have enormous inflows and relatively small volumes. Fluctuation in lake levels has dramatic impacts on the internal and surrounding ecosystems and despite management efforts, is largely regulated by seasonal and annual (e.g., persistent) weather patterns. Groundwater reserves in the Great Lakes are present in multiple locations and at many different depths and flow rates. Besides their role in providing drinking water for numerous communities, they provide important base-flow release to keep the Lakes' feeder rivers perennial and flowing year-round.

Wetlands in the Great Lakes region are extensive and are represented by four distinct types classified as swamps, marshes, bogs, and fens. Swamps are distinguished by the

presence of trees and shrubs and significant annual water cover. Marshes are known for their thick stands of aquatic plants rather than trees and like swamps, are most likely located in the southern Lake region. Bogs are most well known for their stagnant acidic water and slowly decaying, thick mats of sphagnum mosses; fens are filled by slowly moving, less acidic groundwater, dominated by sedges and grasses, but may have stunted shrubs and trees. Many of these wetlands are located along the coastal area, including the UNESCO Biosphere Reserve at Long Point on the north shore of Lake Erie.

Weather patterns and their long-term average condition known as climate have dynamic annual fluctuations in temperature and precipitation due to the Lakes' middle latitude location, which is a global meeting area for continental polar and maritime tropical air masses pushed by the jet stream. January mean daily air temperature ranges from -20 to 0°C , and July mean daily air temperature ranges from 7.5 to 25°C . Mean annual precipitation ranges from 600 to 1300 mm, and snowfall ranges from 150 to 350 mm. The mean annual frost-free period ranges from 220 days to a period less than 2 months north of Lake Superior. Rossby long waves perturbed by short-wave upper airflow patterns trigger the deepening of low-pressure systems, termed cyclogenesis, which results in open wave cyclones defined by the overrunning of warm fronts by cold fronts. Several such open wave cyclones can cover the Great Lakes region, and depending on the season, this can result in synoptic snowfall and rainfall as the cold, dry air mixes with the saturated Gulf Coast air.

Several other weather patterns are also familiar to the region. When the semipermanent Bermuda high-pressure system moves off the southeastern coast after autumn's first frost, the area may experience an Indian Summer as warmer Gulf Coast air is pushed north. Winter weather patterns common to this region include the severe lake effect snows, caused by cold air warming and absorbing lake water, and then cooling on the leeward shore and precipitating, and the northeastern (Nor'easter) winds caused by a strong high-pressure system in Quebec that bring Atlantic Ocean moisture into the region, with ice storms and wet snows. Spring weather is famous for its oscillation between extreme heat and cold, again as a result of the dynamic middle latitude looping of the Gulf Stream that can drag very warm spring Gulf Coast air northward or extremely cold, spring Hudson Bay air southward. Summer weather in the Great Lakes is moderated from extreme heat by the lake temperatures, which may set up lake-land breezes,

Table 2. Water Budgets for Great Lakes Water System ($\text{m}^3 \text{s}^{-1}$)

Category	Superior	Michigan	Huron	Erie	Ontario
Runoff to lake	14,000	11,000	14,000	7,000	9,000
Precipitation to lake	21,000	15,000	15,000	7,000	5,000
Evaporation from lake	15,000	10,000	13,000	7,000	3,000
Flow in by diversions	2,000	-1,000	0	-2,000	2,000
Flow in from upper lake	0	0	37,000	53,000	58,000
Flow out to lower lake/river	22,000	15,000	53,000	58,000	71,000
Retention time (yr)	191	99	22	2.6	6

as well as provide water vapor for convective afternoon thundershowers.

LAND AND WATER USE

Settlement in the Great Lakes extends back to Native American occupation following the Wisconsin icesheet retreat 10,000 years ago; the estimated population during the sixteenth century was between 60,000 and 117,000. The famous Iroquois confederacy of six tribes (Cayuga, Mohawk, Onondaga, Oneida, Seneca, and Tuscarora) occupied much of the New York lands draining into Lake Ontario. European settlers arrived in the early seventeenth century; boats were initially limited to the St. Lawrence Valley due to severe rapids. By 1680, French fur trading in Canada was well established and protected by army forts; by 1730, the British Fort Oswego protected colonies along the southern shore of Lake Ontario. The British captured many French forts by 1759, and the British retained control of the Great Lakes during the American Revolution, granting lands to nearly 40,000 loyalists who fled New England. The 2-year War of 1812 between America and Britain for the Great Lakes was claimed a victory by both sides; America secured the southern coast, and Canada retained its northern control.

Approximately 1 million people in the western Great Lakes region, who sowed crops, worked dairy, and built canals to ship exports and bring new products, firmly established an agricultural heritage for the region by the mid-1800s. Wetlands were drained, soils eroded by denuded lands, rivers were clogged by sediment, and many fish spawning habitats were destroyed. Soil conservation measures have helped stabilize soils, and chemical fertilizers, pesticides, herbicides and irrigation have been recommended for increasing production. Chemicals such as excess phosphorous and historic use of DDT, along with modern row crop monoculture and hormone disrupting chemicals, have significantly impacted the ecosystem. Agricultural production brings vital income and needed food to the region and occupies between 3 and 67% of basin land and upward of 33% of shoreline land, as reported in Table 3.

Logging and forestry occupy even greater amounts of land, from 21 to 91% in the basin (see Table 3), and

commercially began in the 1830s by targeting the light and strong white pine, desired for ship and building construction. Forest industrial uses, primarily pulp and papermaking, but also tannins, created additional revenue and development of the region. Again, however, several negative environmental consequences of this industry, such as water pollution and extensive area, high grade, clear cutting have since been identified. Now it operates at lower volumes and uses best management practices to target sustainable operations. Commercial fisheries were also of major importance; in the late 1890s, harvests had grown by 20% per year since 1820, and annual harvests of 65,000 tonnes were recorded. Today, lake trout, sturgeon, and lake herring have largely been replaced by introduced species of smelt, alewife, splake, and Pacific salmon; average harvests have been around 50,000 tonnes. Recreation boating, swimming, and ecotourism along with sport fishing also represent significant uses and income on the Lakes.

Industrial and commercial development cultivated along the shores of the Great Lakes provided waterpower and waterways for shipping, through which in the 1820s, the Erie Canal and Welland Canal were built. By 1960, water level controls allowed operation of the St. Lawrence Seaway, and the region's principal commodities, iron ore, coal, and grain, were heavily shipped between lakes and to the Atlantic Ocean. Industrial manufacturing, aggressively developed in this region, fostered the growth of many important cities such as Duluth, Marquette, Chicago, Detroit, Toledo, Hamilton, Cleveland, and some towns that had less waterborne commerce such as Rochester, Buffalo, and Toronto in Lake Ontario. Polluted discharges from these industrial centers caused a significant negative impact on the Great Lakes, further discussed in the next section. Episodes of cholera in 1854 and typhoid fever in 1891 were the impetus to reverse the flow of the sewage-spoiled Chicago River away from Lake Michigan, a drinking water source; Hamilton, faced with contaminated lakeshore water, installed deeper intakes into Lake Ontario.

Aquatic ecosystem resources and dynamics are extensive and complex. Driving the food web is sunlight, supplying energy for autotrophs such as phytoplankton, algae, and plants, which in turn feed the heterotrophs, from zooplankton to herbivore fish, which then feed the larger ecosystem carnivore and omnivore predators. Fish species for this region include alewife*, Atlantic salmon, bloaters, bowfin, brook trout, brown trout*, burbot, carp*, Chinook salmon*, Coho salmon*, freshwater drum, lake herring, lake sturgeon, lake trout, lake whitefish, longnose sucker, muskellunge, northern pike, pink salmon*, pumpkinseed, rainbow smelt*, rainbow trout*, rock bass, round goby*, round whitefish, ruffe*, sea lamprey*, smallmouth bass, walleye, white bass, white perch*, white sucker, and yellow perch, where * indicates invasive species. This process continues upward to the amphibian, avian, and terrestrial herbivores and predators and is repeated in many adjoining wetlands and rivers. Popular avian and terrestrial animals in this region include cormorants, gulls, waterfowl, eagles, snapping turtles, whitetailed deer, moose,

Table 3. Basin and Shoreline Land Use Type (%)

Category	Superior	Michigan	Huron	Erie	Ontario
<i>Basin Land Use</i>					
Agricultural	3	44	27	67	39
Residential	1	9	2	10	7
Forest	91	41	68	21	49
Other	5	6	3	1	5
<i>Shoreline Land Use</i>					
Agricultural	n/a	20	15	14	33
Residential	n/a	39	42	45	40
Recreational	n/a	24	4	13	12
Commercial	n/a	12	32	12	8
Other	n/a	5	7	16	7

beaver, muskrat, fox, wolf, and black bear. Decomposition returns the organic nutrients to the mineral soil and inorganic system, an important process carried out by bacteria and fungal organisms. Natural and anthropogenic supplies of essential micro- and macronutrients are critical for the proper biological functioning of this food web, but the introduction of metals, industrial chemicals, and excess nutrients has caused problems, including bioaccumulation and biomagnification, some of them are discussed in the next section.

WATER RESOURCE MANAGEMENT

Pioneering international water resources management has been forged along the shores of the Great Lakes, possibly a challenge best met by two countries that have relatively stable political and economic systems. In 1909, the Boundary Waters Treaty established binational guidance on regulating water levels and flows, which have been the responsibility of the International Joint Commission (IJC), an agency of and for Canada and the United States interests. Separate legislation in 1972, called the Great Lakes Water Quality Agreement (GLWQA), established guidance on regulating water quality, again overseen by the IJC and implemented by numerous national, provincial, state, and small government agencies. Dams are operated on the St. Marys River, Niagara Falls, and the St. Lawrence and only directly affect Lake Superior water levels. These few control points prevent the IJC from dampening all fluctuation; municipal, manufacturing, and power production withdrawals (see Table 4) are affected by and affect levels. Weather exerts a larger control on water level variability, but the IJC has concluded that costs to engineer controls outweigh benefits. Beginning in 1973, the IJC has included erosion control within its water level management objectives of hydropower and navigation, and today the IJC has established new water level guidance to allow greater fluctuation, principally in Lake Ontario, for ecosystem health.

Water quality management under the GLWQA, signed in 1978, began with an analysis of the mass balance loading of phosphorous and other pollutants, leading the way for major subsequent advances in setting target loading limits and current total maximum daily load analysis.

Table 4. Municipal, Manufacturing, and Power Production Water Withdrawals ($10^6 \text{ m}^3 \text{ yr}^{-1}$)

Category	Superior	Michigan	Huron	Erie	Ontario
<i>Withdrawals</i>					
Municipal	98	2,622	384	2,685	927
Manufacturing	1,133	8,608	2,158	9,820	2,935
Power production	740	12,131	4,852	12,791	13,282
<i>Consumptive Use</i>					
Municipal	18	169	170	189	152
Manufacturing	71	785	89	1,409	125
Power production	9	214	62	178	174

Nutrients and carcinogens have received the greatest research and management focus, but the initial analysis of hormone disrupting chemicals in the Great Lakes food chain has identified new concerns and represents a growing issue. An ecosystem approach was adopted by the GLWQA, which was implemented through more expansive Lakewide Management Plans (LaMPs), but areas of concern that contained extreme pollution were also targeted by Remedial Action Plans (RAPs). Innovative water resource management methods developed by the IJC and national agencies such as the Environmental Protection Agency and Environment Canada that encouraged citizen involvement in seeking and implementing solutions have since been studied and shared within the UNESCO Hydrology for the Environment Life and Policy (HELP) Initiative, a needs driven hydrology policy-management-science program to which Lake Ontario belongs. Advanced hydrologic research, such as remote sensing driven modeling and monitoring, coupled with consensus based watershed management, will ideally answer pressing water resource issues.

New management issues have replaced the traditional water chemistry concerns and include the response to several invasive species. Of the many invasives, including crustaceans, fish, mollusks, and plants, some new introductions such as the spiny water flea and zebra muscle are wreaking havoc on industrial to recreational uses of the Great Lakes. Ballast dewatering operations of commercial shipping vessels have been identified as the major pathway for invasives, and U.S. Coast Guard regulations prohibiting this operation are in place. Other challenges include the restoration of coastal wetlands; many of them were destroyed by the removal of extreme peak and low water levels that are responsible for the killing back of monocrop grasses that outcompete the more complex ecosystems needed for proper fish spawning. Restoration of urban streams, including the separation of storm and sanitary sewers, will also be required to regain healthy tributary and headwater spawning areas for Lake species. Additionally greater management control is needed for agricultural, suburban, and urban management of nonpoint source runoff, currently the biggest source of sediment, nutrient, metal, organic, and chemical pollutant loading, now that point source discharges have largely been controlled. A final note must address the recent climate change analysis performed as part of the U.S. Global Change Research Program, which predicts a most likely scenario of lower lake levels due to the establishment of greater evaporation rates, which would potentially have devastating impacts on many recreational, commercial, industrial, navigational, and other Great Lakes activities.

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GREENHOUSE GAS EMISSIONS FROM HYDROELECTRIC RESERVOIRS

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The presence of certain trace gases (e.g., CO₂ and CH₄) in the atmosphere is related to human activity (fossil fuel combustion, deforestation, and intensive agriculture), and to the biogeochemical processes that occur in natural environments (tropical and boreal forests and tundra). For many years now, scientists have been trying to determine the importance of natural environments as sinks for or sources of these trace gases (1–3). The terrestrial biosphere constitutes a major carbon sink (4); more than a third of the anthropic CO₂ emissions are fixed there (5). However, certain changes in land use (e.g., deforestation and the draining of marshes for agriculture) can lead to changes in the way greenhouse gases (GHGs) are produced or fixed by modifying the physicochemical characteristics of these soils. In the medium and long term, these changes are likely to invert the carbon sink capacity that has been attributed until now to certain natural environments (e.g., boreal regions). This article examines how one of these land-use changes, the creation of hydroelectric reservoirs, significantly modifies the carbon cycle in natural environments.

STATE OF RESEARCH

The scientific community's interest in GHG production by hydroelectric reservoirs originated, in part, from the controversy launched by the publication of a paper by Rudd et al. (6). In this publication, the authors concluded that "...per unit of energy produced, greenhouse-gas flux to the atmosphere from some hydroelectric reservoirs may be significant compared to greenhouse-gas emission by fossil-fuelled electricity generation...". This conclusion, based on a weak sampling of fluxes and a method of calculation described as inadequate by certain scientists, created keen interest among researchers (7,8) and the scientists affiliated with hydroelectric companies (9–11). Above and beyond the exercise of comparing energy production methods, one fact remains: that hydroelectricity can no longer be described "as clean energy" with respect to GHG emissions (12–17).

Boreal Reservoirs

One of the studies that has contributed greatly to furthering our knowledge focused on boreal reservoirs in central northern Québec (18).

It was initially assumed that CO₂ and CH₄ fluxes measured on the surface of the reservoirs came mainly from the degradation of organic material in the flooded soil (12). The first research endeavors to understand the processes involved were devoted to surface sampling of GHG emission. Parallel to sampling these surface fluxes, certain morphological parameters (depth of the water column, type of flooded soil), physicochemical (temperature

of water), and meteorological data (wind speed, temperature) were also measured to determine their effects on the variability of these fluxes. Over the years, a significant amount of data was collected concerning GHG emissions from hydroelectric reservoirs in northern Québec. These data, as well as complementary work in microbiology, geochemistry, and other scientific fields, have made it possible to understand better the complex series of processes that control the annual carbon balance in the reservoirs. These processes include the production of GHGs through the degradation of flooded organic matter (soil and vegetation), the production/fixation of GHGs by bacteria and plankton in the water column, and the loss of GHGs to the atmosphere by diffusion, convection, and effervescence (production of gas bubbles) (19).

The processes responsible for producing GHGs in reservoirs and, in particular, production from the degradation of organic material in flooded soil are still being studied. Moreover, we can make no definitive qualitative or quantitative statements about the nature and importance of the various processes involved in producing CO₂ and CH₄. The original assumption about the origin of these gases supposed that the flooding that occurs when a hydroelectric reservoir is created starts significant bacterial activity that degrades the flooded terrestrial organic matter. This activity leads to mineralization of the organic matter and the liberation of CO₂ in the water column. There is likewise the creation of anoxic zones in the flooded soils or in the hypolimnion that supports methanogenesis (formation of CH₄). The carbon source at the origin of the reservoirs' GHG emissions is thus located mainly in the flooded soils (12).

A subsequent study by Weissenberger et al. (20) concerning the total quantity of submerged carbon versus the issue rates of CO₂ and CH₄ at the surface of the reservoirs anticipated that after 100 years, the submerged organic matter would be completely degraded. In practice, however, we are far from observing such a phenomenon. The Cabonga reservoir (central northern Québec), which is more than 70 years old, shows few signs of advanced degradation. A scuba diver observed that certain submerged conifers were still intact after being submerged all these years (9). Moreover, one statistical study by Duchemin et al. (17) showed that there was little difference in CH₄ fluxes for reservoirs of different ages and that there was only a very slight decrease (less than 1% per year) of CO₂ fluxes according to the age of the reservoirs. According to Weissenberger and collaborators (20), the degradation of organic material in the flooded soil cannot solely explain CO₂ and CH₄ fluxes observed at the surface of the reservoirs.

Migrating carbon (from the drainage basin) in transition in the reservoir seems to constitute another considerable source of the carbon involved in GHG production. In fact, new avenues of research suggest that GHG production initially comes from the degradation of organic material in the flooded soil. After a few years, GHG production seems to remain stable due to a major change in the trophic state of the reservoir. Reservoirs go from an autotrophic state (self-sufficient in terms of energy; photosynthetic production exceeds the loss due to respiration)

to a heterotrophic or autotrophic state (dependency on an external source of energy, where photosynthetic production is lower than respiration). This change leads to the assimilation of migrating, allochthonous carbon by living organisms in the reservoir. The bacteria are probably responsible for transforming this carbon into GHGs, and the process seems to occur in the entire water column. All of this is still hypothetical and constitutes a new research project in itself (21).

With regard to the bacterial activity in the water column, experiments in microbiology by Duchemin and his collaborators allowed them to observe that a significant portion of the CH_4 was oxidized before it even reached the reservoir surface (12). The CH_4 is oxidized in the first 25–50 centimeters above the flooded soil. The rate of oxidation is significant in flooded peat bogs and very low in flooded podzols. According to Duchemin et al. (19), two conditions are necessary for methane oxidation: a dissolved oxygen concentration higher than 1 mg L^{-1} and a dissolved inorganic nitrogen concentration lower than $20 \text{ }\mu\text{M}$. This process is at work mainly during the ice-free season because, the weak dissolved oxygen concentrations generated an anoxic zone in the winter which largely reduced methane oxidation. However, at the present time, very few observations are available to quantify the process of methane oxidation adequately.

Planktonic activity did not appear to be a determining factor in the GHG concentration of the hydroelectric reservoirs of central northern Québec. Though biological activity (fixation-respiration) had suggested that the maximum dissolved CO_2 concentration occurred at night, a study by Canuel et al. (1997) has shown that the maximum concentration actually occurs during the day. The authors thus deduced that the evolution of the GHG concentrations in the reservoir is controlled by processes other than planktonic activity.

Conversely, the processes responsible for transporting dissolved gases in the water column are relatively well understood. It is assumed that most of the transport occurs vertically, that is, there is little horizontal current and homothermy. The vertical transport of dissolved gases results from turbulent diffusion and convection. The turbulent diffusion is induced by mechanical forcing (e.g., wind) which acts in a stable density gradient. For its part, the convection occurs following density anomalies that cause an unstable profile. This can ensue from the deep absorption of solar radiation or from the loss of sensible heat or latent heat through evaporation or through ice melting. These two transport mechanisms (turbulent diffusion and convection) vary in intensity according to the period of the year and weather surface conditions. For example, in autumn, major convective movements are responsible for transporting a significant quantity of dissolved gas toward the reservoir surface due to the static instability generated by surface cooling (23). According to Duchemin and Levesque (24), effervescence (i.e., release by bubbles) constitutes another transfer mechanism of GHGs in a reservoir. However, the contribution of this mechanism is only significant for CH_4 in the very shallow parts of reservoirs (24). For CO_2 , it is estimated that this

transport mechanism contributes less than 1% of the total annual emissions (25).

Kelly et al. (26) conducted a study in northwest Ontario in which GHG emissions were measured before and after the controlled flooding of a peat bog and pond. It was observed that the flooding of the peat bog greatly modified the GHG emission balance. Before flooding, the peat bog constituted a CO_2 sink and a weak source of CH_4 . After flooding, it became a significant source of CO_2 and CH_4 . According to Kelly et al. (26), this CO_2 status inversion is partly caused by the death of almost all the vegetation in the natural environment, making it incapable of fixing CO_2 , and by the decomposition of the submerged organic matter. Furthermore, the reinforcement of its status as a CH_4 source is due to an increase in the anoxic decomposition of the vegetation and to a reduction in the consumption of CH_4 (oxidation) by oxidizing bacteria. It would seem that the first mechanism is quantitatively more significant than the second.

Tropical Reservoirs

Reservoirs in tropical regions have also been studied. The first studies indicated that tropical reservoirs constitute even more significant GHG sources than northern reservoirs (15,16,24,27). According to Fearnside (27), the results obtained until now even call into question the idea that hydroelectricity in the Amazon region will produce less GHGs than the production of the same quantity of energy by other fossil fuels. According to Pearce (15), the Amazon basin brings together ideal conditions for producing methane; little dissolved oxygen, relatively high temperatures, and high concentrations of nutrients. This author also states that all the hydroelectric reservoirs of the world produce the equivalent of 7% of the total production of CO_2 . Duchemin et al. (24) point out that Amazonian reservoirs produce ten times more CO_2 and three times more CH_4 than the hydroelectric reservoirs of northern Québec.

CONCLUSION

The biosphere and, in particular, the soils contain substantial carbon reserves. Flooding following the creation of a hydroelectric reservoir can therefore significantly modify this carbon reserve. The boreal regions and their soils have acted as a carbon sink since the last glaciation. This capacity can, however, be reversed. As for the tropical regions, their GHG emissions can be so significant as to call into question hydroelectric development.

From the point of view of the international effort to reduce GHGs, it would appear essential to compare the annual total GHG emissions arising from different energy forms; an exercise that has been carried out by several scientists (28–30). Duchemin (29) has shown that the boreal reservoirs have annual equivalent CO_2 emissions lower than those emitted by power stations fueled by coal or natural gas when producing an equivalent quantity of energy (see Fig. 1). On the other hand, he has also shown that certain tropical reservoirs can emit quantities

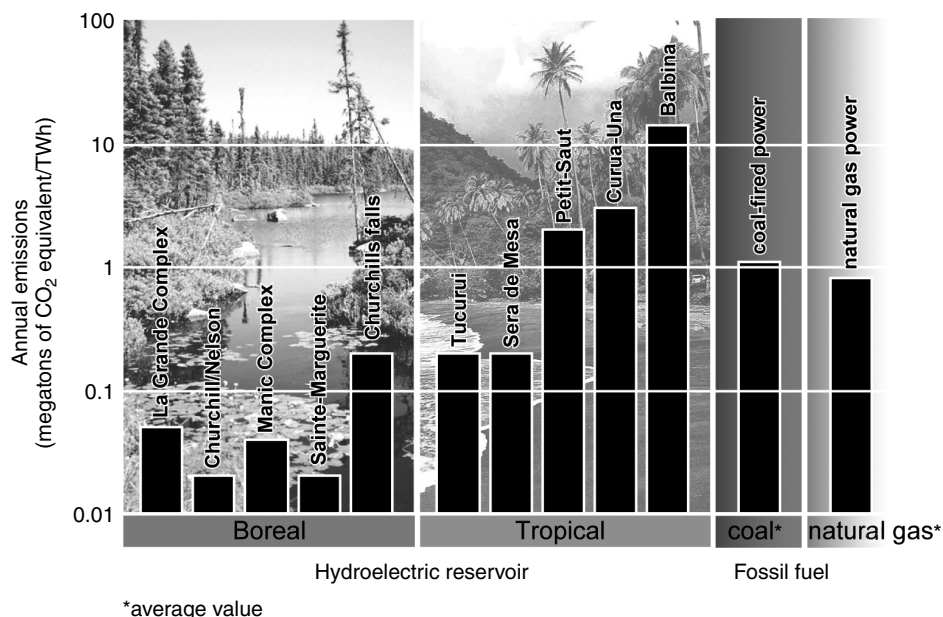


Figure 1. Greenhouse gas emissions from hydropower and fossil fuel power (29).

of equivalent CO₂ up to 14 times higher than the emissions from a coal-fired power station producing a comparable quantity of energy (see Fig. 1). It is particularly worth noting that shallow tropical reservoirs (6 m and less) emit up to two to three times more CO₂ per square meter than deep reservoirs (25 m and more). Thus, the site for constructing a hydroelectric reservoir in a tropical area can be a determining factor in its emission balance. The creation of hydroelectric reservoirs in zones of fairly flat topography should be avoided because the flooded surface areas will be substantial and will provide ideal conditions for developing the bacteria responsible for these emissions.

Duchemin insists that the current measurements of emissions in tropical reservoirs are still too limited to offer a clear answer to the question of energy choices. Moreover, he adds that future studies should introduce the concept of the net before-and-after-flooding balance to evaluate truly the changes in the carbon sink or source capacity of a given territory.

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GULLY EROSION

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Gullies have formed rapidly in many parts of the world during the last 300 years following the change from

traditional agriculture or nomadic indigenous lifestyles to more intensive European-based agriculture. They are deeper erosion channels than rills, but their differentiation is often unclear and arbitrary because of a continuum between the two features. Rills are shallow, ephemeral erosion channels that can be obliterated by standard tillage, alternating cycles of freeze-thaw, or rain-impacted shallow overland flows.

DEFINITION

Brice (1) proposed that gullies are recently extended drainage channels that transmit ephemeral flow, have steep sides, a width > 0.3 m, a depth > 0.6 m, and terminate at a steeply sloping or vertical, upstream primary nickpoint (see next section for definition of within-gully features). Gregory and Walling (2) concluded that gullies are characterized by

- ephemeral streamflow,
- steep sidewalls and nickpoints,
- a depth greater than rills so that they cannot be eliminated by plowing, crossed by a wheeled vehicle, or restored in the short term,
- a V-shaped cross section in fine-grained, resistant subsoils,
- a U-shaped cross section where the soil and subsoil are equally erodible, and
- unconsolidated sediments in the sidewalls.

Attempts to differentiate between gullies and streams on the basis of flow frequency are difficult for two reasons. First, gullies are not restricted to specific climatic regions but have been described in arid, semiarid, subhumid, and humid areas (11). Second, flow regimes vary systematically based on drainage basin area and climate, (31) and gullying is not restricted to specifically sized basins nor to areas of ephemeral flow.

Brice (1) adopted a depth > 0.6 m as representative of a gully, but a depth of 0.3 m has been proposed by soil conservationists (3) and a depth of 0.5 m by Prosser and Winchester (4). Gullies need not be defined by the exceedance of a critical depth because the depth of incision, cross-sectional shape and longitudinal profile of gullies are all dependent on the materials into which the gully is eroded (5–7).

To overcome the above problems, the definition proposed by Schumm et al. (8) is adopted. A gully is a relatively deep, recently formed eroding channel that forms on valley sides and on valley floors where no well-defined channel previously existed. In contradistinction, incised streams form by the erosion of a preexisting river channel (8). Valley-side gullies result in an expansion of the drainage network due to erosion of soil, colluvium, and/or bedrock in hillside hollows. Valley-floor gullies may be either continuous or discontinuous and erode valley-fills of alluvium, swamp deposits, and/or bedrock. Arroyos, which have been the subject of much research in the American southwest, are valley-floor gullies that have

steeply sloping or vertical walls of cohesive, fine sediments and flat and generally sandy floors (9).

GULLY EROSION PROCESSES

Gully erosion occurs by a number of erosional processes involving nickpoint or nickzone initiation and upstream retreat (Fig. 1), downstream progressing degradation, and sidewall erosion. Nickpoints are an abrupt change in bed profile that involve a substantial, local increase in slope and retreat by migrating upstream at a rate that declines exponentially over time (5,8). Primary nickpoints mark the upslope limit of gully erosion, whereas secondary nickpoints rework sediment temporarily stored in the gully bed (Fig. 1). Nickzones are steep sections at the headward limit of erosion cycles and also migrate upstream over time without maintaining a vertical face (8). The rotation of nickpoints into nickzones usually occurs when the eroded valley fill is not stratified (6) or when it is uniform, noncohesive material (5). During downstream progressing degradation, bed slope is reduced, the bed often becomes armoured or veneered with gravel, if present, and bed material discharge declines, approaching a minimum value consistent with upstream sediment supply (10). During the evolution of gullies, initial upstream nickpoint migration is often replaced by subsequent downstream progressing degradation, as upstream sediment supply declines due to the exhaustion of most of the stored sediment within the gully network. These erosion cycles can be repeated many times. Sidewall or bank erosion

following incision occurs by many processes that operate at different rates (7,11) and can, in some cases, erode more sediment than the initial nickpoint erosion (12).

Gully Sediment Yields

Actively developing gullies generate very high sediment yields that have caused concern to land and river managers throughout the world. However, rates of gully erosion are not included in equations for standard erosion prediction (13). The author's data on sediment yields from gullies of different age in southeastern Australia have been combined with that of Prosser and Winchester (4) to produce Fig. 2. Erskine and Saynor's (14) detailed review of the available sediment yield data for Australian rivers showed that most rivers have yields less than $50 \text{ tkm}^{-2}\text{y}^{-1}$, and many are also less than $10 \text{ tkm}^{-2}\text{y}^{-1}$. High yields in excess of $120 \text{ tkm}^{-2}\text{y}^{-1}$ have been found only for basins that have been extensively disturbed by agriculture or significantly gullied. Figure 2 demonstrates that developing gullies less than 5 years old produce sediment yields greater than $2000 \text{ tkm}^{-2}\text{y}^{-1}$, but that these high yields rapidly decline during the next 20 years. Williams (15) reported the very high sediment yield of $93,750 \text{ tkm}^{-2}\text{y}^{-1}$ for a small developing gully in the seasonally wet tropics of northern Australia, where sediment yields are usually very low by world standards (8). Neil and Fogarty (17) used farm dams to determine sediment yields from continuous and discontinuous gullies in comparison to native forest in southeastern Australia. They found that discontinuous gullies, on average,

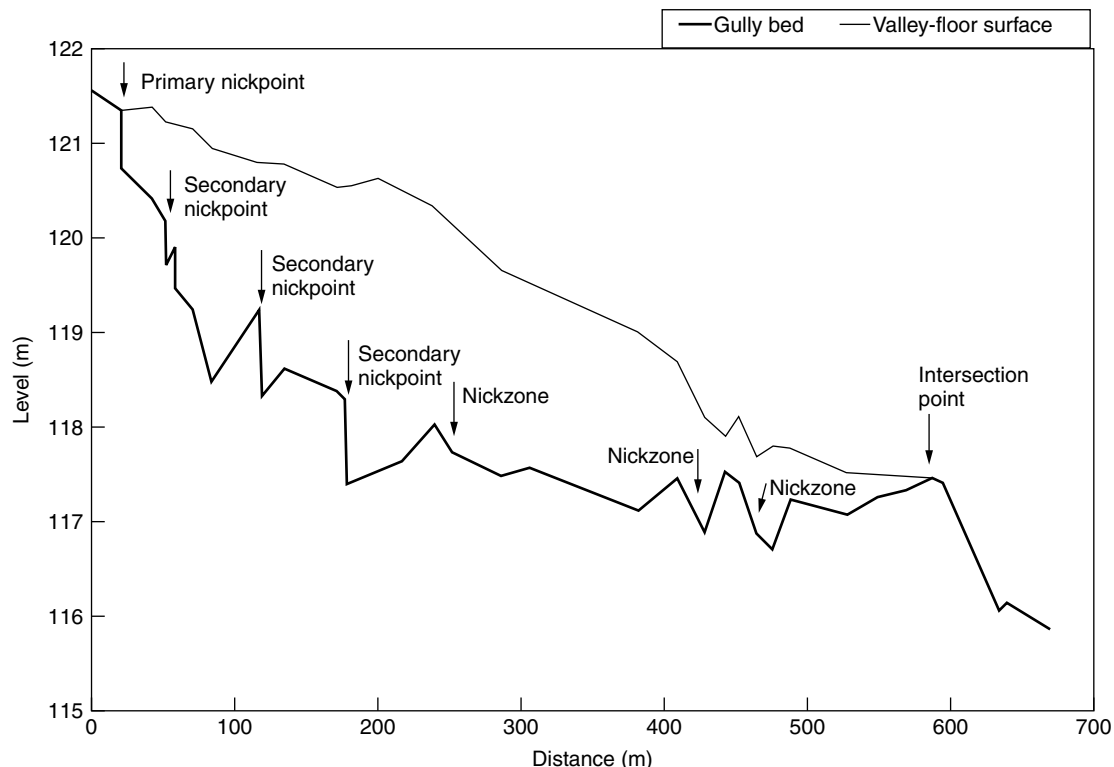


Figure 1. Longitudinal profile of the Dry Arm discontinuous valley-floor gully in the upper Wollombi Brook drainage basin in southeastern Australia, showing some of the main features.

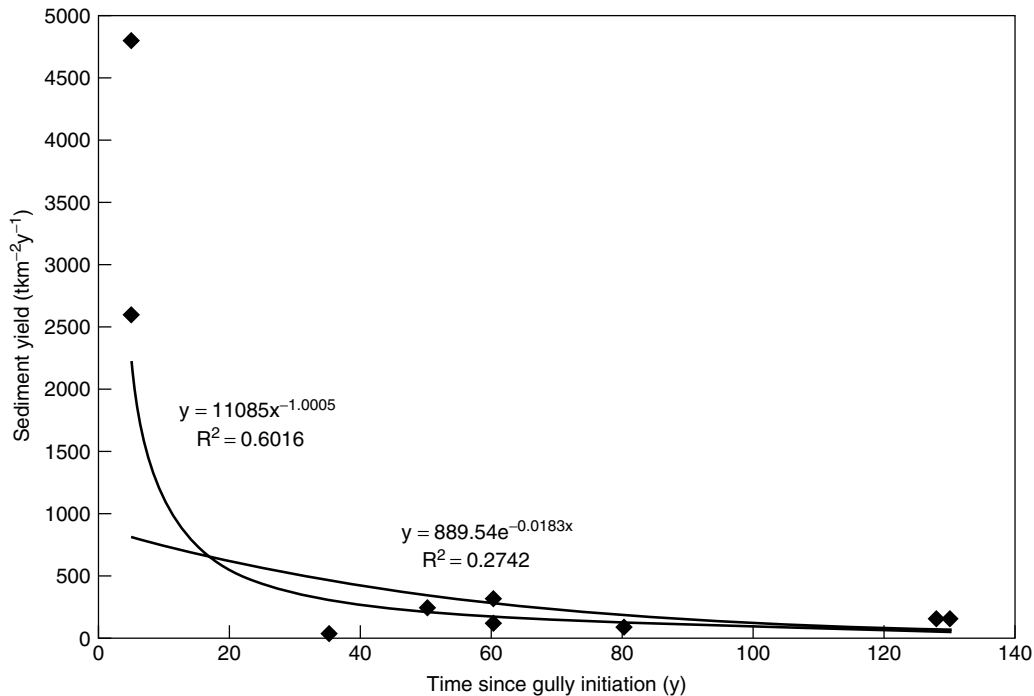


Figure 2. Variations in gully-derived sediment yields with time since gully initiation in southeastern Australia.

produced more than 11 times higher yields than native forests and that continuous gullies generated the highest yields at more than 73 times greater than native forests. Graf (18) demonstrated that an exponential decline in rates of gully erosion occurs in Colorado, consistent with relaxation following initial gully erosion. However, for the data in Fig. 2, a power function is a much better fit than an exponential function. Nevertheless, there is a rapid decline in sediment yield following the first 20 years after gully initiation.

Not all of the sediment generated by gully erosion is transported directly into higher order streams because significant amounts are often stored within the gully and immediately downstream. Melville and Erskine (19) found from detailed field measurements on a compound discontinuous gully system draining a 13.8 km² basin in southeastern Australia that 90% of the 190,000 m³ eroded since European settlement was trapped in the gully bed and small floodplain within the gully as well as in the fan immediately downstream, without ever being exported from the basin. More recent research by the author on other discontinuous gullies in southeastern Australia accounted for all of the gully-eroded sediment which was entirely stored within the gully and the downstream fan. Similarly, Neil and Fogarty (17) estimated that as much as 60% of the material eroded from discontinuous gullies in southeastern Australia was deposited solely in the downstream fan and hence was not transported out of the basin.

Sediment movement in drainage basins is often restricted to clearly defined compartments that may be up to tens of kilometers long but are not linked to higher order streams (19,20). Therefore, gully erosion often results in reworking sediment from one temporary storage to

another without a detectable impact on the sediment yields of large drainage basins (21).

CAUSES OF GULLY EROSION

Cooke and Reeves (9) proposed a deductive model of arroyo formation which maintains that a gully is initiated when the erosion potential of flows (erosivity) locally exceeds the resistance of the surficial materials (erodibility). Therefore, there are two fundamental changes, increased erosivity and increased erodibility, which either together or individually, can lead to gully initiation (9). Erosivity refers to the propensity for flows to detach and remove materials from a given area more rapidly than they are replaced from upstream and is a function of many interrelated hydraulic, channel form, and sediment load variables that are difficult to evaluate precisely (9). Nevertheless, recent research has demonstrated that grass is very important in increasing critical boundary shear stress for soil erosion (22,23). Erodibility is a measure of surficial sediment's resistance to erosion forces and is determined by soil texture, soil aggregation, soil coherence to wetting, soil organic matter content, and vegetative cover. However, there are many trigger mechanisms that can potentially lead to gully formation; they have been classified (Fig. 3) as natural geomorphic processes, gradational geomorphic thresholds, random climatic events, secular rainfall changes, and land use changes (9,24).

Figure 3 shows an expanded version of Cooke and Reeves's deductive model without the explicit links between specific trigger mechanisms and the induced changes in erosivity and/or erodibility. The reason for the omission of these explicit links is to show an uncluttered

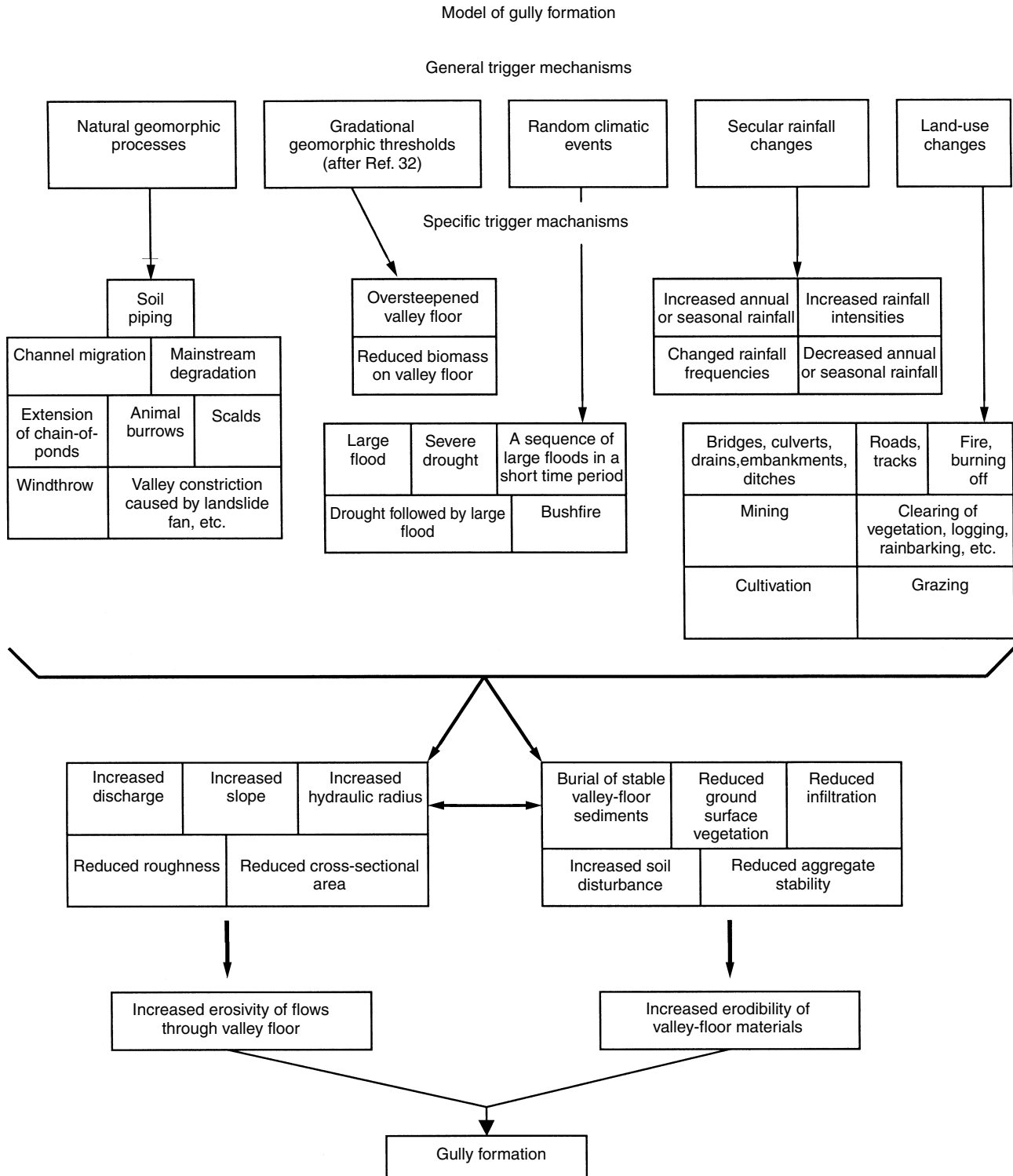


Figure 3. Deductive model of gully formation (modified from Refs. 9 and 24).

hierarchical representation of the interrelationships. Examples of the possible specific trigger mechanisms are outlined in the second tier, and the way they effect erosivity and erodibility is listed in the third tier of Fig. 3.

It is important to emphasize that the results of detailed investigations of gully initiation have found that

apparently similar gullies can be formed in different areas as a result of different combinations of initial conditions and specific trigger mechanisms (9). For example, the author found that in upper Wollombi Brook, a 341 km² basin in humid southeastern Australia, for the period since first European settlement in the 1820s,

- the timing of valley-floor gully initiation was highly variable, ranging over at least 100 years and only rarely coincided with the period of peak agricultural productivity in the middle of the nineteenth century,
- most gullies were initiated at various times following long periods of sediment storage in valley segments without inflowing tributaries that eventually increased valley-floor slope until it exceeded a stability threshold (gradational geomorphic threshold in Fig. 3),
- some gullies were initiated by a catastrophic flood when up to 508 mm of rainfall were recorded in 12 hours in June 1949 (25) (random climatic event in Fig. 3),
- some gullies were initiated at various times by culvert construction, drains, and ditches in localized parts of valley floors (19) (land use changes in Fig. 3), and
- many valley floors remained ungullied despite extensive valley-floor forest clearing, high intensity agricultural activities involving repeated cultivation, extensive valley-floor roading, repeated high intensity wildfires, and intensive rural subdivision.

Figure 4 depicts the least-squares linear regression relationship between field-surveyed, mean valley-floor slope and basin area for 19 sites (nine ungullied and 10 gullied) in the upper Wollombi Brook basin. For the gullied sites, the valley-floor slope of the presumed initiation point of the gully was surveyed. As this is frequently buried by sand eroded from the upstream gully, multiple excavations were conducted to reveal the original valley-floor surface which was readily identified using the techniques outlined by Happ et al. (26). A locally steeper slope segment was found at the downstream end of each gully,

as first reported elsewhere (27). The slope and constant of the regression equation are consistent with reported values elsewhere (13). The regression line effects a clear separation between gullied and ungullied sites (only one gullied site plots below and no ungullied sites plot above the line) and represents a gradational threshold slope for the initiation of gullying in this basin. Similar results have been reported elsewhere (8,13,27,28).

EFFECTS OF GULLY EROSION

The environmental, human, and economic effects of gully erosion documented by many authors (for examples, see Refs. 7–9, 13, 24, 26, 28–30) include all of the following:

1. high soil erosion rates and sediment yields
2. substantial loss of productive farmland
3. deterioration of downstream water quality
4. rapid and massive downstream sedimentation
5. replacement of a relatively stable, muddy substrate by very mobile sand
6. more peaked flood hydrographs
7. higher flow velocities due to decreased resistance to flow and greater flow depths
8. reduced baseflow persistence
9. reduced frequency of overbank flow
10. lower floodplain water tables
11. loss of valley-floor wetlands and seepage zones and their associated vegetation
12. reduced macroinvertebrate, reptile, amphibian and fish biodiversity and some local extirpations by the wholesale loss of aquatic habitats
13. severe farm access and management problems

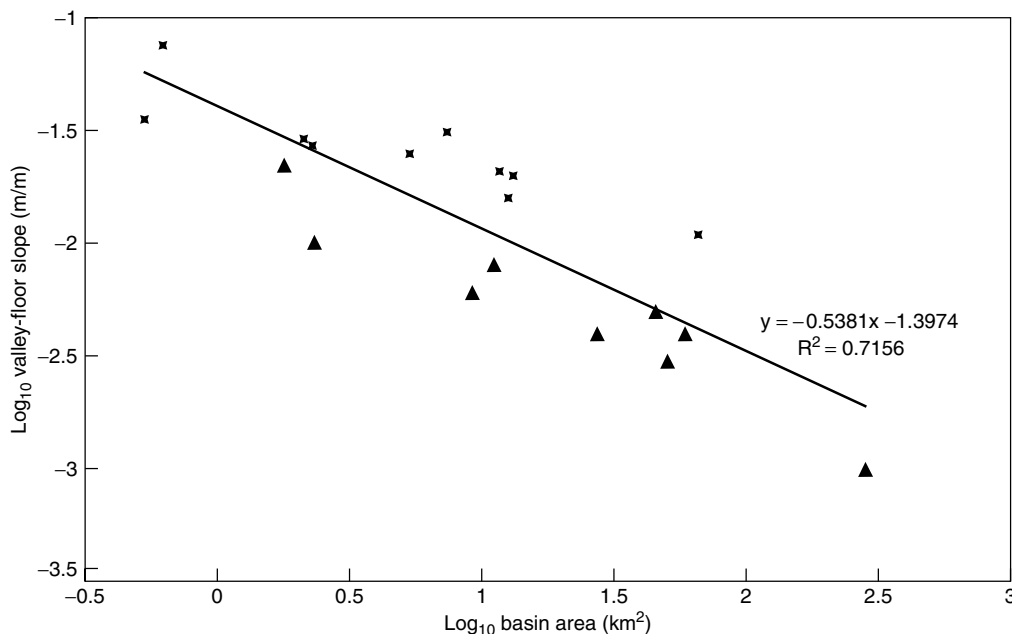


Figure 4. Bivariate plot of mean valley-floor slope versus drainage basin area for various gullied and ungullied sites in the upper Wollombi Brook drainage basin, southeastern Australia. The gullied sites are represented by crosses and the ungullied sites by triangles.

14. flooding of houses by soil-laden water
15. reduced farm incomes due to a decline in land productivity
16. disruption and dislocation of transport routes
17. damage to, and loss of, human structures, such as bridges, pipelines, pump sites and water intakes.

These effects have been recorded at the site of initial gully erosion as well as upstream and downstream (8,9).

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POTENTIAL HEALTH ISSUES ASSOCIATED WITH BLUE-GREEN ALGAE BLOOMS IN IMPOUNDMENTS, PONDS AND LAKES

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INTRODUCTION

Cyanobacteria, also known as blue-green algae, are the most primitive group of algae. Although they are most closely related to other bacteria, they have the

same pigment for photosynthesis that other algae and plants have. They are simple but remarkably successful organisms. Individually, they are microscopic; however, large clusters of cells are easily visible as a surface scum (a type of algal bloom) on any stable body of water. Some kinds of blue-green algae produce natural toxins. Unfortunately, no visual technique can distinguish between toxin-producing blue-greens and those that are harmless. Ingestion of these toxins has caused the poisoning of domestic animals, sometimes resulting in death. These include cattle, horses, sheep, hogs, ducks, dogs, and wildlife. Human exposure and toxicity, ranging from intestinal problems to liver damage and fatalities, has only recently been documented.

TOXINS TYPES

The toxins are generally of two types: hepatotoxins and neurotoxins. Hepatotoxins affect the liver, disrupting the important proteins that keep the liver functioning. They generally act more slowly, and a higher dose is needed before death occurs; however, they also may be tumor promoters at low doses. Microcystins, the most common and important family of hepatotoxins, have over 60 variations of a basic cyclic peptide structure. Most of these are hepatotoxins, however, their toxicity depends on the specific amino acids that are part of the basic structure (Fig. 1). Related compounds include nodularin, nodulapeptins, anabaenopeptins, and aeruginopeptin. Another increasingly important hepatotoxin is cylindrospermopsin.

Neurotoxins are normally fast-acting; ingestion of a large dose causes paralysis of skeletal and respiratory muscles which results in death. These compounds are potent alkaloids rather than cyclic peptides. The most common forms are anatoxin, anatoxin-a(S), saxitoxin, neosaxitoxin, and related compounds (Fig. 2).

All of these toxins are difficult to identify and quantify and require detailed laboratory analysis using techniques such as liquid chromatography and mass spectroscopy. Canada, Australia, and Great Britain have developed a guideline level of 1 microgram microcystin per liter of

water, or 1 part per billion (1 ppb) for drinking water, and this standard has been adopted by the World Health Organization. During algal blooms, toxin levels can greatly exceed 1 ppb.

SOURCES OF TOXINS

Hundreds of kinds of blue-green algae are known from aquatic habitats; however, only a handful of these are of concern. Some common ones that are known toxin producers include species of *Aphanizomenon*, *Microcystis*, and *Anabaena* (Fig. 3). Others include species of *Planktothrix* (*Oscillatoria*), *Nostoc*, *Anabaenopsis*, *Nodularia*, *Cylindrospermum*, *Cylindrospermopsis*, and *Lyngbya*.

Blue-green algae blooms may appear like thick pea soup or grass clippings on the water and are most common in late summer, although blooms can happen in spring or year-round in the warmest latitudes. Pigments and toxins are produced inside the cells and stay there as long as they are alive. When cells break down, usually when a bloom begins to die, their pigments are released into the water and may look like green or blue paint. Bloom die-off also leads to oxygen depletion and associated fish kills in a waterbody. This results from bacterial decomposition of these dead algal cells, which consumes the oxygen. Generally, cooler weather, rainfall, and reduced sunshine will lead to the collapse of an algal bloom. Some blooms last a few weeks; others persist for a few months, depending on environmental conditions. Nutrient enrichment often increases the amount of blue-green algae and may also enhance the dominance of noxious forms.

Control of Blue-Green Algal Blooms

Reducing the load of nutrients from the watershed and/or preventing the release of nutrients from bottom sediments are the primary control measures. In most systems, phosphorus is the critical nutrient. Because in-lake controls treat mostly symptoms, reducing the input of nutrients from the watershed is preferable.

In-lake controls include physical methods such as aeration, circulation, dilution, flushing, and light-limiting

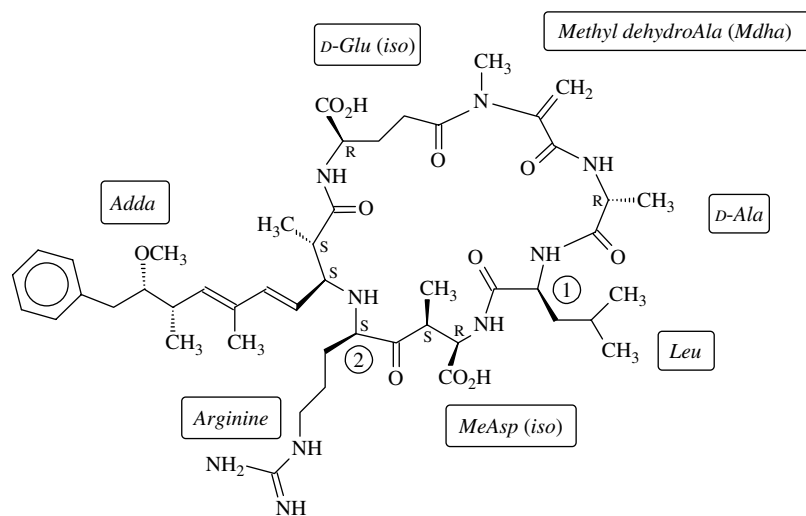


Figure 1. The generic structure of a microcystin. Amino acids substituted at positions 1 and 2 account for most of the variations in this family of compounds (e.g., microcystin-LR contains the amino acids leucine (L) and arginine (R) at positions 1 and 2, respectively) (courtesy of Dr. Gregory Boyer, SUNY-Syracuse).

Figure 2. Structures of the alkaloid cyanobacterial toxins: anatoxin-a, anatoxin-a(S), saxitoxin, and cylindrospermopsin. Anatoxin-a, -a(S), and saxitoxin are neurotoxins. Cylindrospermopsin is a hepatotoxin. Saxitoxin is a representative of a large toxin family referred to as the paralytic shellfish poisons (courtesy of Dr. Gregory Boyer, SUNY-Syracuse).

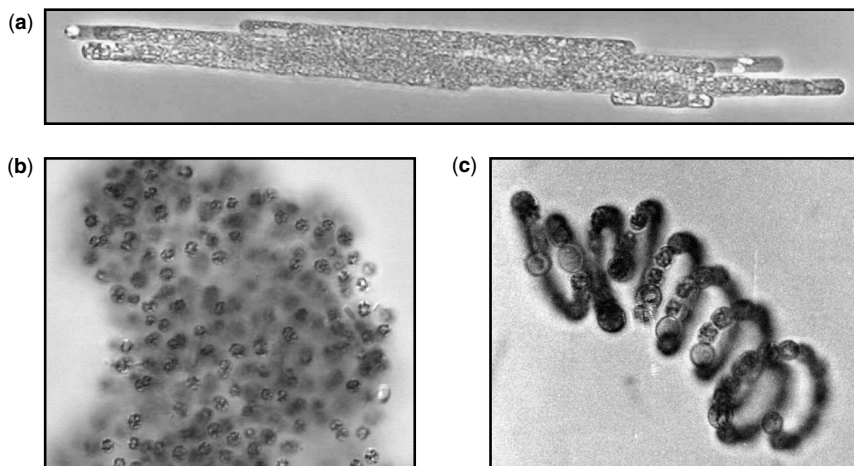
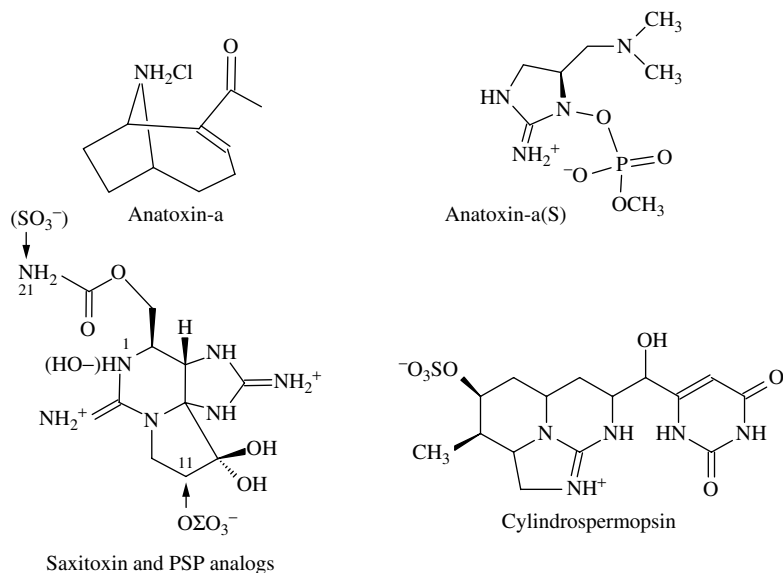


Figure 3. Known toxin producing blue-green algae as seen through a microscope. (a) *Aphanizomenon flos-aquae* (b) *Microcystis aeruginosa* (c) *Anabaena circinalis*.

dyes. Chemical controls include algicides and phosphorus inactivation. Biological controls include in-lake plantings and barley straw. All of these techniques have advantages and disadvantages that need to be addressed for controlling an algal bloom.

HEAT BALANCE OF OPEN WATERBODIES

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INTRODUCTION

From a continuum-mechanical point of view, the heat balance or the thermodynamics of a lake, reservoir, or ocean is governed by three different types of processes: surface fluxes, those through the water surface and the sediment boundary of a water body; internal heat

production; and external heat supply. Whereas surface fluxes are accounted for in boundary conditions, supply and production take place in the water volume and are described as additional production terms in the equation of heat (1). Turbulent diffusion and, to a much lesser degree, molecular diffusion lead to a certain temperature stratification, usually dividing a lake into a well-mixed, upper warm layer, the epilimnion, separated from it by a transition zone a colder hypolimnion. The temperature-depth distribution can be determined from the heat balance equation and boundary conditions describing the surface fluxes into the water. In contrast to this, in a well-mixed waterbody with a homogeneous temperature distribution, for example, a polymictic lake, all heat production and flux terms are lumped together in a net thermal energy flux. In this case, the heat production terms are treated as fluxes across the surface. Using such a simple system, one can determine the equilibrium temperature of the waterbody for a certain set of meteorologic conditions, that temperature to which the mixed waterbody is driven under the same prevailing

meteorologic conditions. For a stratified lake or ocean, such an approach is only of limited use because it does not account for vertical transport of heat in the waterbody. But it can be a good approximation for the surface temperature when the stratification is strong, thus separating the hypolimnion from the well-mixed epilimnion.

The heat budget of a waterbody is determined mainly by the heat exchange across the water surface through long-wave (infrared) radiation, evaporation, and conduction, plus heat production through the absorption of short-wave radiation in the water column. To a far less extent, inflows (rivers, discharges) and outflows, precipitation, heat conduction across the bottom, and biogeochemical reactions alter the heat balance. These effects become relevant only in certain cases: in/outflow in reservoirs and lakes with short residence time (2); heavy precipitation by thunderstorms (3); sedimentary (or geothermal) heat flow in transparent shallow lakes (4); or in amictic or meromictic lakes, where the water body is decoupled from the atmosphere (5).

Special conditions occur in ice-covered lakes. Here, heat conduction and short-wave absorption through the snow and ice cover limit the heat entering the waterbody. The parameters describing these processes depend strongly on factors such as the type of ice, ice thickness, water inclusions, snow cover, and snow type (6). The description of the heat transfer across the air–ice interface down into the lake water thus needs to include additional formulations for snow and ice (7).

HEAT BALANCE

Meteorologic Driving Factors

Heat exchange across a water surface is related to the meteorologic conditions above the water surface and the water surface temperature itself. The main driving factors are air temperature, water vapor pressure, and wind speed above the water surface. They control conductive and evaporative heat transfer. Radiative heat fluxes are additionally influenced by clouds and atmospheric conditions. Usually only a limited set of meteorologic variables are known for calculating heat fluxes using empirical relations. These variables comprise air temperature T_a and relative humidity H , usually measured 2 m above ground level, wind speed U , cloud cover C , and global irradiance I . Instead of relative humidity, dew point temperature or water vapor pressure might be given. Wind speed drops significantly when measured nearer to the ground, so it has to be scaled to a fixed level, for example, 10 m above ground, using a logarithmic velocity profile depending on the roughness height of the terrain (8).

General Heat Balance Formulation

The main components of the net heat exchange Q_{net} between atmosphere and water can be divided into radiative and nonradiative terms. Radiative heat fluxes are distinguished according to their wavelengths. Short-wave radiation Q_s entering the water column has its major spectral parts in the range from 400–700 nm,

whereas long-wave, infrared radiation is in a range of approximately 10 μm . In contrast to short-wave radiation, which penetrates the water column, infrared radiation from clouds and the atmosphere, Q_{lin} , is absorbed directly at the water surface. Depending on the surface temperature T_w of the waterbody, it behaves as a so-called blackbody and also emits infrared radiation, Q_{lout} . Nonradiative heat flux terms are due to evaporation or condensation Q_e and the sensible heat flux Q_c generated by conduction.

Omitting other less important heat fluxes such as geothermal heat flux or those by in/outflows, the general heat balance is given by

$$Q_{\text{net}} = Q_s + Q_{\text{lin}} + Q_{\text{lout}} + Q_e + Q_c \quad (1)$$

The magnitudes of the different heat fluxes are shown in Fig. 1 for Lake Nieuwe Meer in The Netherlands.

Empirical Relations

The relations between meteorologic fields and heat fluxes are evaluated by empirical formulas. Usually they are derived from measurements at lakes or the ocean in certain climatic and/or geographic regions. One can find a multitude of parametric values for a certain empirical relation for different waterbodies. Thus, those relations cannot be generalized, and care must be taken in applying them to other settings. Collections of different formulations and parametric sets are described in the literature (e.g., 9–11). In the following, one set of these empirical relations is shown that exemplifies their structure. The heat flow direction is assumed here to be positive from the atmosphere into the waterbody.

Short-Wave Radiation. Solar irradiance at the water surface is partly reflected, depending on the solar zenith angle. The zenith angle Θ itself depends on latitude λ , the day of the year d , and daytime τ . In addition, the reflectivity r of the water depends on wave action, which

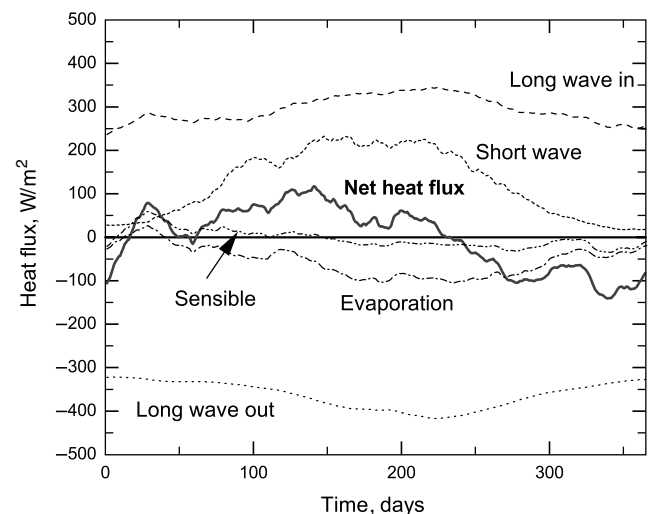


Figure 1. Heat fluxes for Lake Nieuwe Meer, The Netherlands, during the year 2002 (30-day running mean).

can be parameterized by the wind speed U across the water surface (12). If the global irradiance incident at the surface Q_{irr} is measured, the irradiance entering the water column is given by $Q_s = Q_{\text{irr}}[1 - r(\Theta, U)]$. If the irradiance is unknown, it can be calculated from the solar constant S and the angle of incident light at the location (13,14): $Q_{\text{irr}} = S \cos \Theta$. Obviously, this is an upper bound for a clear sky and must be modified for cloud cover by multiplying by an attenuation function $g_s(C)$ (9).

Short-wave radiation that penetrates into a waterbody decays with depth. In its simplest form, this process is described by the Lambert–Beer law: exponential decay $\exp(-kz)$ with depth z and vertical extinction coefficient k . More sophisticated models describing light extinction with depth that depend on water constituents, phytoplankton biomass, and spectral composition of the light can be found in Reference 14. In a well-mixed water column, the depth distribution of heat absorption does not affect the heat balance, as long as there is no significant heat flux into the sediment.

A simple formulation for heat flux into the water column generated by short-wave radiation is given below. Some effects such as the dependence of solar radiation on the Sun–earth distance, atmospheric attenuation, or local effects, for example, shading, are not included here.

Formulation. Earth's declination [day of year: d (days)]:

$$\delta = 23.45 \cdot \frac{2\pi}{360} \cdot \sin \left[\frac{2\pi}{365.25} (d + 283) \right] \text{ (deg)} \quad (2)$$

Zenith angle [latitude: λ (deg); time: τ (hours)]:

$$\Theta = \arccos \left[\sin \delta \sin \lambda - \cos \delta \cos \lambda \cos \left(\frac{2\pi}{24} \tau \right) \right] \text{ (deg)} \quad (3)$$

Refractive index air–water:

$$n_w = 1.33 \quad (4)$$

Snell's law [incidence angle: Θ ; refraction angle: Θ_w (deg)]:

$$\sin \Theta_w = \frac{\sin \Theta}{n_w} \quad (5)$$

Reflection coefficient:

$$r = \frac{1}{2} \left\{ \left[\frac{\tan(\Theta - \Theta_w)}{\tan(\Theta + \Theta_w)} \right]^2 + \left[\frac{\sin(\Theta + \Theta_w)}{\sin(\Theta - \Theta_w)} \right]^2 \right\} (-) \quad (6)$$

Reflection coefficient depending on wind speed for large zenith angles (wind speed: U [m/s]):

$$r = 0.09[4.444 \exp(-0.17U) + 2.977] \left(\frac{180}{\pi} \Theta - 65.5 \right)^{0.040816} (-), \quad \Theta > 65.5^\circ \quad (7)$$

Effect of cloudiness [Cloudiness: C (–) range [0,1]]:

$$g_s = 1 - 0.65 C^2(-) \quad (8)$$

Solar constant (for the mean distance between the Sun and the earth)

$$S = 1353 \text{ W/m}^2 \quad (9)$$

Incident irradiance:

$$Q_{\text{irr}} = S \cos \Theta \text{ (W/m}^2\text{)} \quad (10)$$

Heat flux of short-wave radiation entering the water column:

$$Q_s = (1 - r)g_s Q_{\text{irr}} \text{ (W/m}^2\text{)} \quad (11)$$

Long-Wave Radiation from the Atmosphere and Clouds. Long-wave radiation from the atmosphere, clouds, and the water surface can be treated as blackbody radiation; thus, according to the Stefan–Boltzmann law, it is proportional to the fourth power of the body's surface temperature. For long-wave radiation into a lake, cloud cover increases the heat flux, which is taken into account by a quadratic expression for the cloudiness $g_l(C)$. The proportionality factor, emissivity ε_a , depends on a number of factors, such as air temperature, vapor pressure, and cloud height. A common expression for it is given (see e.g., Ref. 9) as a quadratic function of air temperature.

Formulation. Emissivity of the atmosphere [air temperature: T_a (°C)]:

$$\varepsilon_a = 0.919 \cdot 10^{-5} (T_a + 273)^2 (-) \quad (12)$$

Effect of cloudiness [cloudiness: C (–) range (0, 1)]:

$$g_l = 1 + 0.17 C^2(-) \quad (13)$$

Stefan–Boltzmann constant:

$$\sigma = 5.6697 \cdot 10^{-8} \text{ W m}^{-2} \text{ K}^{-4} \quad (14)$$

Heat flux of long-wave radiation from the atmosphere and clouds:

$$Q_{\text{lin}} = \varepsilon_a \sigma g_l (T_a + 273)^4 \text{ (W/m}^2\text{)} \quad (15)$$

Long-Wave Radiation from a Water Surface. The emission of infrared radiation from a water surface takes the usual form for blackbody radiation at the temperature of the water surface, where the emissivity ε_w of water is constant. It yields the Stefan–Boltzmann law for long-wave radiation from a water surface, given below.

Formulation. Emissivity of water:

$$\varepsilon_w = 0.97(-) \quad (16)$$

Heat flux of long-wave radiation from a water surface [water temperature: T_w (°C)]:

$$Q_{\text{lout}} = -\varepsilon_w \sigma (T_w + 273)^4 \text{ (W/m}^2\text{)} \quad (17)$$

Evaporation. Evaporation from a water surface, a heat loss term in evaporation and a source term for condensation, depends on the water temperature and the water

vapor pressure above the water surface e_a (see, e.g., Reference 15). The latent heat of evaporation/condensation multiplied by the water's density $\rho_w L(T_w)$ describes the energy needed to evaporate a given volume of water. The amount of water per unit surface, the evaporation height, which evaporates per time, E , depends on the atmospheric conditions above the water column. It is proportional to the gradient of water vapor pressure, which can be approximated by the difference in saturated vapor pressure at ambient water temperature $e^{(\text{sat})}(T_w)$ and the vapor pressure above the water column. The latter can be computed as the saturated vapor pressure at the dew point temperature T_{dew} or via the relative humidity H defined as the ratio of actual to saturated vapor pressure, depending on the meteorologic data available. The effect of wind speed on the evaporation $f_{\text{wind}}(U)$ mixing the air above the water is often assumed to have a linear form (9,16). As humidity is increasing when moving downwind across a water surface, evaporation decreases slightly with the size of a lake (or the wind fetch). To correct for that effect, an additional factor $R_{\text{size}}(A)$ dependent on lake size is applied (17,18).

Formulation. Latent heat of evaporation/condensation [air temperature: T_w (°C)]:

$$L = 2.5 \cdot 10^6 + 2365T_w \text{ (J/kg)} \quad (18)$$

Saturated vapor pressure from the Clausius–Clapeyron equation [temperature: T (°C)]:

$$e^{(\text{sat})}(T) = 1230 \exp \left[5362.97 \left(\frac{1}{283} - \frac{1}{T + 273} \right) \right] \text{ (Pa)} \quad (19)$$

Water vapor pressure in the air [dew point temperature: T_{dew} (°C); rel. humidity: H (–)]:

$$e_a = e^{(\text{sat})}(T_{\text{dew}}) = H e^{(\text{sat})}(T_a) \quad (20)$$

Wind function [wind speed: U (m/s)]:

$$f_{\text{wind}} = 1 + 0.59U \text{ (–)} \quad (21)$$

Lake size scaling [surface area: A (km²)]:

$$R_{\text{size}} = \left(\frac{5}{A} \right)^{0.05} \text{ (–)} \quad (22)$$

Evaporation rate:

$$E = 1.36 \cdot 10^{-11} f_{\text{wind}} R_{\text{size}} [e^{(\text{sat})}(T_w) - e_a] \text{ (m/s)} \quad (23)$$

Density of water:

$$\rho_w = 1000 \text{ kg/m}^3 \quad (24)$$

Heat flux by evaporation/condensation:

$$Q_e = -\rho_w L E \text{ (W/m}^2\text{)} \quad (25)$$

Sensible Heat. The sensible heat flux across a water surface depends on the temperature difference between the water and the overlying air. Assuming that the

transport of heat in the atmosphere is similar to that of water vapor, the sensible heat flux due to conduction and convection can be modeled in parallel to the latent heat flux. The Bowen ratio B (19,20) relating these two fluxes can be approximated by the ratio of temperature differences and water vapor pressure differences across the surface and a linear dependence on air pressure p_a . Air pressure is either measured or approximated with a barometric height formulation to adapt for different elevations of the water surface.

Formulation. Bowen ratio [atmospheric pressure: p_a (Pa); air and water temperatures: T_a, T_w (°C)]

$$B = 0.61 \cdot 10^{-3} p_a \frac{T_w - T_a}{e^{(\text{sat})}(T_w) - e_a} \text{ (–)} \quad (26)$$

Sensible heat flux:

$$Q_c = B Q_e \text{ (W/m}^2\text{)} \quad (27)$$

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HYDRAULICS

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INTRODUCTION

Hydraulics is the branch of physics that studies the equilibrium (hydrostatic) and the motion (hydrodynamics) of fluids. Fluids can be divided into gases and liquids. Hydraulics treats the liquid that generally are practically incompressible and occupies definite volumes.

THE PROPERTIES OF LIQUIDS

The density ρ is the mass per unit volume, and it is influenced both by the temperature and the pressure. For water at 4 °C, $\rho = 1000 \text{ kg/m}^3$. The specific weight γ is the weight of a unit volume of substance. For water, $\gamma = 9.813 \text{ kN/m}^3$. The viscosity μ is the properties of a fluid that determine the resistance to a shearing force. For liquids, it decreases as the temperature increases, and for water at 20 °C, it is $1.14 \cdot 10^{-3} \text{ kg/(m s)}$. The surface tension is because of the molecular attraction and is responsible for the curvature of the fluid surface near the wall of vessels.

HYDROSTATIC FORCES

A liquid at rest in a vessel exerts a pressure on the base and on the wall. This pressure is called hydrostatic. If A is the area of the base, the force per unit area exerted is $p = W/A$, where W is the weight of the liquid in the vessel. Further, the pressure p applied to any horizontal plane passing through the liquid is $p = \gamma h$, where h is the vertical distance of the plane below the free surface (Stevin's law). Besides, any external pressure applied to a fluid is transmitted undiminished throughout the liquid and

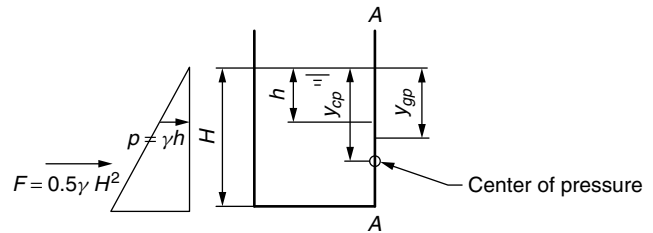


Figure 1. Pressure and resulting forces on the wall A–A of a vessel containing a liquid.

onto the walls of the containing vessel (Pascal's principle), and the pressure at a given depth does not depend on the shape of the vessel containing the liquid or the amount of liquid in the vessel.

In a liquid at rest, viscous and shearing forces do not exist. The vertical distance h between any selected points in a liquid at rest and the free surface exposed to the atmosphere is called the head of the liquid.

The line of action of the resultant force passes through the center of pressure. This center of pressure lies below the centroid, which is an intuitive result, because pressure increases with depth.

It can be determined graphically or analytically using the following formula:

$$y_{cp} = \frac{I_{cg}}{y_{cg}A} + y_{cg} \quad (1)$$

where I_{cg} is the moment of inertia of the area A about its center of gravity axis and y_{cg} is the depth of the center of gravity below free surface (Fig. 1).

The horizontal component of the resulting force acting on any surface is equal to the normal force on the vertical projection of the surface and acts on the center of pressure for the vertical projection. Instead, the vertical component is equal to the weight of the liquid above the area, real or imaginary.

FLUID FLOWS

Fluid flow can be steady or unsteady, uniform or nonuniform. Steady flows occur if the velocity, pressure, and other fluid variables (density, viscosity) are constant with time. Uniform flow instead occurs when no variation in space exists. The two fundamental equations of fluid flows are the equation of continuity and the Bernoulli equation (or the momentum equation). The first expresses the principle of conservation of mass, and in steady conditions and for incompressible fluids becomes

$$A_1 V_1 = A_2 V_2 \quad (2)$$

where A_i and V_i are, respectively, the cross-sectional area and the average velocity at the generic section i .

The Bernoulli equation results from the application of the principle of conservation of energy, and in the direction of the flow is expressed as follows:

$$\frac{p_1}{\gamma} + z_1 + \frac{V_1^2}{2g} = \frac{p_2}{\gamma} + z_2 + \frac{V_2^2}{2g} + \Delta h \quad (3)$$

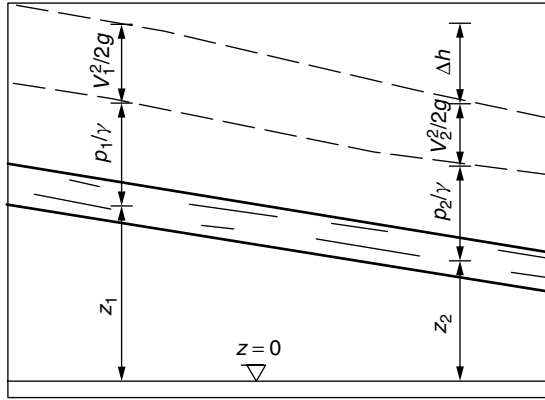


Figure 2. Graphical representation of Bernoulli equation.

where z_i is the height of the fluid above a reference level, p_i the pressure, V_i the velocity, g the acceleration because of gravity, and Dh the energy lost between the two sections. The term p/γ represents the pressure head, z the geometric head, and $V^2/2g$ the velocity head, and their sum is the total energy. The graphical representation of the Bernoulli equation is illustrated in (Fig. 2).

FLOWS THROUGH ORIFICES AND WEIRS

For a tank with a hole with sharp edges (Fig. 3a) flowing in the atmosphere, the discharge Q through the orifice of area A can be obtained applying Bernoulli's equation:

$$Q = \mu A \sqrt{2gh} \quad (4)$$

where h is the water depth above the center of the orifice and μ is a coefficient that takes into account the effect of friction and the contraction of the flow and is equal to 0.61–0.62.

A weir is a wall built in a channel cross section, over which the water flows from one level to another in a controlled way (Fig. 3b). The relationship that links the characteristics of a rectangular weir to the discharge, in the hypothesis that the velocity of the fluid approaching is small so that kinetic energy can be neglected, is:

$$Q = \mu' B h \sqrt{2gh} \quad (5)$$

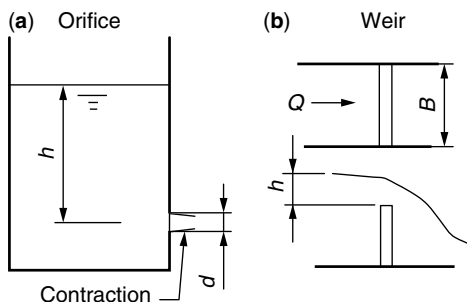


Figure 3. Flow through orifices (a); and weir (b).

where B is the width of the weir, h is the hydraulic depth over it, and μ' is a coefficient (0.42 in case of sharp edge and 0.385 for a wide-crested weir).

FLOWS IN PIPES

Flow in pipes can be laminar or turbulent (1,2). In the first case, the fluids move in parallel layers. In the second one, the particles move in all directions, causing a complete mixing of the fluid. If the flow is laminar or turbulent, it can be evaluated with the Reynolds number, which expresses the ratio between inertial and viscosity forces:

$$Re = \frac{\rho V d}{\mu} \quad (6)$$

where d is the diameter of the pipe. When $Re < 2000$, flow is in laminar condition, whereas for values greater than 2000, it starts to be turbulent.

If a flow is laminar, the head losses Dh can be determined using the Hagen–Poiselle formula:

$$\Delta h = \frac{64 L}{Re d} \frac{V^2}{2g} \quad (7)$$

where L is the distance between two sections.

For turbulent flow, the lost head can be given by means of the Darcy–Weisbach formula:

$$\Delta h = f \frac{L}{d} \frac{V^2}{2g} \quad (8)$$

where f is the friction factor, which depends on the values of Re and on the roughness of the wall of the pipe. According to Colebrook and White, f is expressed as:

$$\frac{1}{\sqrt{f}} = -2 \log \left[\frac{(\varepsilon/d)}{3.71} + 2.51 \frac{Re}{\sqrt{f}} \right] \quad (9)$$

where ε is the dimension of the roughness.

The head losses can also be because of singularity inserted in the pipes (curves, valves, entrance and exiting connections, etc.), and they are expressed as

$$\Delta h = k \frac{V^2}{2g} \quad (10)$$

where k is a coefficient depending on the characteristics of the singularity.

OPEN CHANNEL FLOWS

An open channel is a conduit in which the liquid flows with a free surface subjected to atmospheric pressure (3). For steady and uniform flow, the energy grade line is parallel to the liquid surface (which represents the hydraulic grade line). Define with S the slope of the channel and with R the hydraulic radius, the velocity of the flow can be determined with the Chezy formula:

$$V = C \sqrt{RS} \quad (11)$$

where C is a coefficient depending on the wall roughness determined with the following expression (Bazin):

$$C = \frac{87}{1 + m\sqrt{R}} \quad (12)$$

The coefficient m is given in Table 1.

The hydraulic radius represents the ratio between the wetted area and the wetted perimeter. For a rectangular channel, $R = Bh/(B + 2h)$ where B is the width and h the water depth.

Another expression to determine the hydraulic characteristic of open channel flow is the Manning equation:

$$V = \frac{1}{n} R^{2/3} \sqrt{S} \quad (13)$$

where n is the Manning coefficient (Table 2).

In open channel flow, the specific energy can be introduced as

$$E = y + \frac{V^2}{2g} \quad (14)$$

It represents the sum of water depth and velocity head, and in terms of unit discharge $q = Q/B$, it is

$$E = y + \frac{q^2}{2gy^2} \quad (15)$$

For a constant value of q and for rectangular channel, the specific energy has a minimum for a water depth

$$y_c = \sqrt[3]{\frac{q^2}{g}} \quad (16)$$

This water depth is called critical depth. For a given value of specific energy, it corresponds to the maximum unit discharge.

Table 1. Values for Bazin's Roughness Coefficient

Type of Channel	Bazin's m
Cement: planed wood	0.109
Planks: brick, cut stones	0.290
Rubble masonry	0.833
Earthen channel: very regular	1.54
Earthen channel: in ordinary condition	2.35
Earthen channel: very rough or weed grown	3.17

Table 2. Values for Manning Coefficient

Type of Channel	n
Natural stream: Clean and Straight	0.030
Natural stream: Major Rivers	0.035
Natural stream: Sluggish with Deep Pools	0.040
Excavated Earth Channels: Clean	0.022
Excavated Earth Channels: Gravelly	0.025
Excavated Earth Channels: Weddy	0.030
Floodplains: Pasture, Farmland	0.035
Smooth Steel	0.012
Smooth Steel	0.022

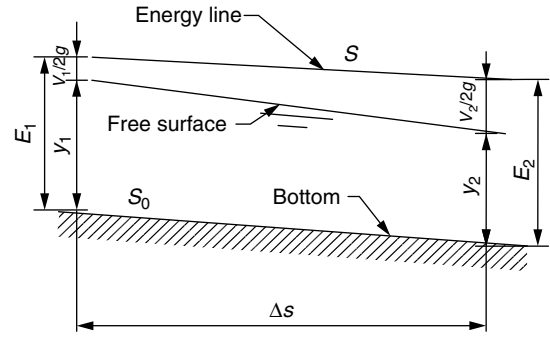


Figure 4. Nonuniform flow condition.

Flows occurring at depth below and above the critical depth are classified as supercritical and subcritical, respectively.

In steady, nonuniform flow conditions, the water depth varies with the distance. These conditions can usually be studied using Bernoulli's theorem and dividing the problem into lengths of approximately uniform condition. For each reach of length ΔS , the energy equation becomes

$$\Delta S = \frac{E_2 - E_1}{S_0 - S} \quad (17)$$

where E_2 and E_1 are the specific energies downstream and upstream, respectively, S_0 is the bottom slope, and S is the slope of the energy line (Fig. 4).

CONCLUSION

A compacted summary of the fundamental principles of hydraulics is presented in this paper. Aspects of hydrostatic and of the motion of fluids have been described from an engineering point of view.

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HYDRAULICS OF PRESSURIZED FLOW

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Hydraulics is a branch of fluid mechanics specifically dealing with water as a major fluid in many engineering systems. It is safe to consider water as an incompressible fluid for typical engineering applications. Pressurized flow usually refers to the flow in closed conduits. Hydraulics of pressurized flow involves three basic equations: continuity, energy, and momentum equations. Head loss is important

in hydraulics of pressurized flow regardless of system design or flow analysis. Darcy-Weisbach and Hazen and Williams equations typically estimate friction head loss. The friction factor depends on Reynolds number and relative roughness of the pipe. Total head loss includes friction head loss caused by fluid viscosity and local head losses at changes in the cross section, bends, and various pipe fittings. A pump provides energy head to, and a water turbine takes energy head from, the pipe system.

Fluid mechanics is the study of behavior of liquids and gasses at rest or in motion subjected to various forces. Hydraulics is a branch of the fluid mechanics specifically dealing with water as a major fluid in many engineering systems. The way water behaves under various conditions depends primarily on its fundamental chemical and physical properties. The bulk modulus of water, which is a measure of its elasticity (or volume reduction under pressure), is about $2.2 \times 10^9 \text{ N/m}^2$ under normal pressure. Therefore, it is safe to consider water as an incompressible fluid for practical purposes. An incompressible fluid does not change its density (ρ) with respect to pressure; therefore, it is also independent of location if no changes of temperature occur in the study area. Pressurized flow usually refers to the flow in closed conduits, e.g., in a water supply pipe system, where the flow is under pressure and full-pipe flow (Fig. 1), whereas pressure could be above or under the local atmospheric pressure.

Hydraulics of pressurized flow involves three basic equations: continuity, energy, and momentum equations. The equation of continuity is a mathematical statement of the Law of Conservation of Mass (water). Along a conduit, inflow rate at the upstream cross section 1 (Q_1) equals the outflow rate at the downstream cross section 2 (Q_2):

$$Q_1 = Q_2 \quad \text{or} \quad V_1 A_1 = V_2 A_2 \quad (1)$$

where V_1 and V_2 are the average velocities at the cross sections 1 and 2 and A_1 and A_2 are the cross-sectional areas. When the branch pipe system is involved, the equation of continuity states that the sum of inflow equals the sum of outflow in the control volume system. The energy equation for incompressible fluid under no heat exchange can be expressed as

$$Z_1 + \frac{p_1}{\gamma} + \frac{V_1^2}{2g} = Z_2 + \frac{p_2}{\gamma} + \frac{V_2^2}{2g} + HL_{12} + H_p - H_t \quad (2)$$

Each term in Eq. 2 has units of length (e.g., meter or feet). The first three terms ($z, p/\gamma, V^2/2g$) are the energy per unit weight of water caused by its elevation, pressure, and velocity, respectively, and they are called elevation head,

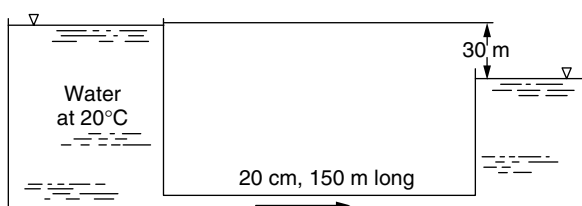


Figure 1. Pressurized pipe flow between two reservoirs.

pressure head, and velocity head. Subscripts 1 and 2 stand for upstream and downstream cross sections 1 and 2, and g is the acceleration of gravity (9.81 m/s^2 or 32.2 ft/s^2). HL_{12} is energy loss per unit weight of water from the cross section 1 to 2 and is called total head loss. H_p and H_t are energy heads provided by a water pump or taken by a water turbine to/from the pressurized flow system:

$$H_p = \frac{P_E \eta}{\gamma Q} \quad \text{or} \quad H_t = \frac{P_T}{\eta \gamma Q} \quad (3)$$

where P_E is the electric power provided to a pump, P_T is the electric power generated by a turbine, η is overall efficiency, γ is specific weight of water, and Q is discharge through a pump or turbine. Some applications may not have pumps or turbine in the system (e.g., in Fig. 1). Head loss is an important term in hydraulics of pressurized flow regardless of system design or flow analysis. Total head loss includes friction head loss caused by fluid viscosity and local head losses at changes in the cross section, bends, and various pipe fittings. Friction head loss (HL_f) is usually determined from the Darcy-Weisbach equation:

$$HL_f = f \frac{L V^2}{D 2g} \quad (4)$$

where f is the friction factor, L is the pipe length between the cross section 1 and 2, D is the pipe diameter, and V is the mean velocity (Q/A). For a noncircular conduit, D is replaced by four times the hydraulic radius that is the flow area divided by wetted perimeter. The friction factor is a function of the Reynolds number (Re) and the relative roughness of a conduit (absolute roughness ε over diameter D). Absolute roughness for concrete, welded-steel and cast iron pipe are 0.3 to 3.0 and 0.12 and 0.25 mm, respectively, and it is a function of pipe material. For laminar flow, f is equal to $64/Re$. For turbulent flow, the Moody diagram presented in many textbooks (1–3) can be used, and Swamee and Jain (4) also suggested an explicit formula in replace of the Moody diagram with restrictions:

$$f = \frac{1.325}{\left[\ln \left(\frac{\varepsilon}{3.7D} + \frac{5.74}{Re^{0.9}} \right) \right]^2} \quad (5)$$

for $10^{-6} < \varepsilon/D < 10^{-2}$ and $5000 < Re < 10^8$. For water supply engineering applications, the Hazen and Williams equation is typically used to estimate friction head loss as

$$HL_f = L \frac{4.727 Q^{1.852}}{C^{1.852} * D^{4.8704}} \quad \text{in English units} \quad (6a)$$

$$HL_f = L \frac{10.675 Q^{1.852}}{C^{1.852} * D^{4.8704}} \quad \text{in the international system of units} \quad (6b)$$

where C is a coefficient given in Table 1 for various types of conduit surface. Local head loss (HL_L) is typically expressed as function of velocity head:

$$HL_L = K \frac{V^2}{2g} \quad (7)$$

Table 1. Values of C for the Hazen–Williams formula [from Khan (1)]

Type of Pipe	C	Type of Pipe	C
Asbestos cement	140	Brass	130–140
Cast Iron		Concrete	
New, unlined	130	Steel forms	140
Old, unlined	40–120	Wooden forms	120
Cement lined	130–150	Centrifugally spun	135
Bitumastic, enamel-lined	140–150		
Tar-coated	115–135		
Copper	130–140	Galvanized iron	120
Lead	130–140	Plastic	140–150
Steel (riveted)	110	Steel (new, unlined)	140–150

Velocity V is typically taken from just downstream of the local fitting. K is the local head loss coefficient and typically determined by experiments. For example, perpendicular square entrance, $K = 0.5$; exit to a reservoir, $K = 1.0$; standard tee entrance, $K = 1.8$; globe valve when fully open, $K = 10$; and close return bend, $K = 2.2$. Many other local head loss coefficients are given in textbooks (1–3). Local head losses from various fittings are also represented as equivalent pipe lengths in many application fields. Therefore, the total head loss can be calculated by Darcy–Weisbach and Hazen–Williams equations alone when length (L) as the sum of equivalent pipe lengths from fittings and actual pipe length is used.

The fluid momentum equation can analyze forces required to keep a pipe bend and a vertical tee in place (Fig. 2). Fluid momentum equation is derived from the Newton's second law of motion as

$$\sum F = \dot{M}_o - \dot{M}_I \quad (8)$$

It means that the sum of all external forces applied on the control volume equals the difference of the rate change of momentum leaving (outflow) and entering (inflow) the control volume. It is easier to arrange the momentum equations in the direction of three rectangular axes x , y , and z .

Energy Eq. 2 can determine what flow rate the pressurized cast iron pipe in Fig. 1 can carry:

$$HL_{12} = f \frac{L}{D} \frac{V^2}{2g} = Z_1 - Z_2 = 30 \text{ m} = f \frac{150}{0.2} \frac{V^2}{2 * 9.81} \quad (9)$$

The cross sections 1 and 2 are located at the surface of upstream and downstream reservoirs, so pressure head and velocity head at the cross sections 1 and 2 are all zero. H_p and H_t are zero as no pump and turbine are in the system. The relative roughness (ε/D) of the cast iron pipe is $0.25 \text{ mm}/20 \text{ cm} = 0.00125$; and by assuming that the Reynolds number is very large first, the friction factor can be estimated from Eq. 5 as 0.02. Therefore, flow velocity V can be calculated from Eq. 9 as 6.26 m/s. Now one needs to recheck the Reynolds number and recalculate friction factor and velocity:

$$Re = \frac{VD}{\nu} = \frac{6.26 \text{ m/s} * 0.2 \text{ m}}{1.007 \times 10^{-6}} = 1.24 \times 10^6 \quad (10a)$$

$$f = \frac{1.325}{\left[\ln \left(\frac{0.25}{3.7 * 200} + \frac{5.74}{(1.24 \times 10^6)^{0.9}} \right) \right]^2} = 0.02 \quad (10b)$$

where ν is the kinematic viscosity of water at 20°C . Because the friction factor is still 0.02, the final velocity is the same as the first trial, and the flow rate in the pipe is

$$Q = VA = 6.26 \text{ m/s} * (\pi/4 * 0.2^2) = 0.196 \text{ m}^3/\text{s} = 6.95 \text{ ft}^3/\text{s} \quad (11)$$

where local head losses (entrance and exit) are relatively small and ignored in the above calculation. For fully turbulent conditions at high Reynolds numbers, discharge for a pressurized pipe flow can be calculated as

$$Q = \left[\frac{\pi}{4} \sqrt{2g} \left(1.14 - 0.86 \ln \frac{\varepsilon}{D} \right) D^{2.5} \right] \left(\frac{HL_f}{L} \right)^{1/2} \quad (12)$$

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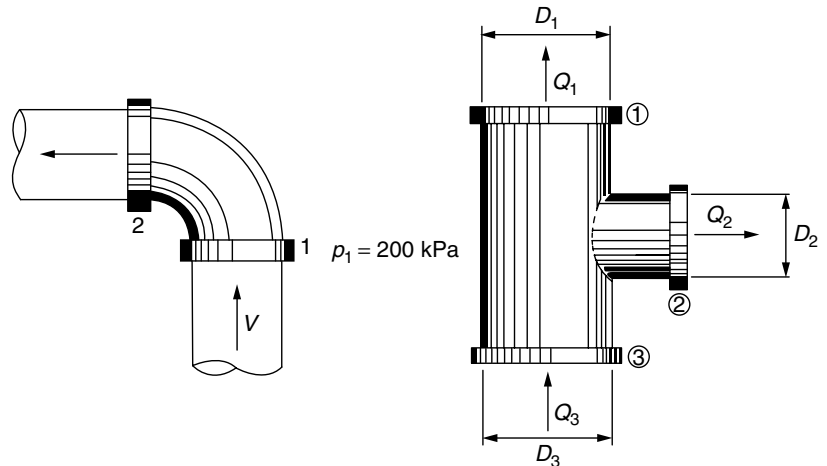


Figure 2. Examples for momentum applications: a pipe bend and a vertical tee.

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HYDROELECTRIC POWER

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INTRODUCTION

Hydropower generation follows a simple concept: water falling from the force of gravity turns the blades of a turbine, which is connected to a generator. The rotating generator produces electricity. Prehistoric man was aware of the energy contained in falling water. One of the earliest devices to use this energy was the water wheel. The Romans used the energy of falling water to do many useful things. They constructed paddle wheels that turned with river flow and lifted water to troughs built higher than river level. The Egyptians and Greeks harnessed the power of river currents to turn wheels and grind grain before 2000 B.C. In the Middle Ages, more efficient water wheels were built for milling grain.

In the nineteenth century, the water turbine gradually replaced the water wheel, and dams were built to control the flow of water. Since then, the hydroelectric potential of rivers continued to be developed. The first modern turbine design was developed in 1849 by James Francis. According to estimates, hydroelectric power production has now risen to 2,000 billion kilowatt hours worldwide.

DAMS AND HYDROPOWER

As a dam holds back water in its reservoir, the water level rises creating a reservoir of potential power. This water is allowed to pass through the turbines of a hydroelectric power plant in conduits called penstocks. The water turns the turbines which are coupled with generators to produce electricity. This electricity passes through a transformer and reaches factories, shops, and homes on transmission lines. The principle and the technique to generate electricity from water remain the same regardless of the size of the project. A plant may serve a small community or a country. For example, many communities in remote areas of Nepal are not connected to the national grid and get electricity from mini/micro hydropower (size <2 MW) plants. The largest hydroelectric complex in the world now is on the Parana River between Paraguay and Brazil. It is known as the Itaipu Dam, and its 18 turbines produce 12,600 megawatts (MW) of electricity. The Itaipu Dam supplies about 80% of the entire electricity demand in Paraguay and 25% of the demand in Brazil. Even this will be surpassed by

the hydropower plant at the Three Gorges Dam (under construction) across the Changjiang River (China) whose installed capacity will be about 18,000 MW!

QUANTITATIVE HYDROPOWER FACTS

Hydropower is the world's most important and cleanest source of renewable energy. It can be seen from Table 1 that nearly 17.5% of the world's electricity is produced by hydropower.

Canada is the world's biggest producer of hydropower, generating 350 TWh/year (nearly 62% of the country's total electricity production). Canada produces more than 13% of the global output of hydropower and is the world's second largest exporter of electricity after France.

Europe and North America have developed more than 60% of their hydropower (see Table 2), most of it during the twentieth century. In contrast, many countries in Asia, South America, and Africa currently use only a small portion of their potential hydropower, and a large hydropower potential still remains unexploited.

The electricity supplied by hydropower far exceeds the capacity of any other renewable energy resource. Norway meets virtually its entire (99.6%) electricity demand by hydropower. Twenty-five countries worldwide depend on hydropower for more than 90% of their electricity needs. Though fossil fuels are dominant in electricity generation, worldwide, more than 60 countries currently use hydropower for half or more of their electricity needs. Most of the installed hydroelectric capacity is in North America, Brazil, Russia, China, and Europe. The amount of hydropower generation by major producers is shown in Table 3.

Table 1. World Electricity Generation by Source, 1999^a

Source	Percentage
Hydropower	17.5
Coal	38.1
Nuclear	17.2
Gas	17.1
Oil	8.5
Other	1.6

^aReference 1.

Table 2. Continentwise Percentage of Hydropower Potential that has been Developed

Continent	Percentage of Hydropower Potential that has been Developed	Percentage of Electricity Generated by Hydropower
Africa	7	2
Asia	20	39
Australia	40	2
Europe	65	13
N. America	61	26
S. America	19	18

Table 3. Hydropower Generation by Major Producers of the World^a

Country	Hydropower Produced TWh	% of World Total
Canada	346	13.0
United States	319	12.0
Brazil	293	11.0
China	204	7.7
Russia	161	6.1
Norway	122	4.6
Japan	96	3.6
India	81	3.0
France	77	2.9
Sweden	72	2.7
Rest of the world	888	33.4
Total World	2659	100.0

^aReference 1.

ADVANTAGES OF HYDROPOWER

Hydroelectric power is a clean, renewable energy source. In contrast to other renewable sources of electricity, hydropower can supply a significant portion of the world's electricity needs, year after year. Because it can respond quickly to changes in electricity demand by controlling the release of water, hydropower can augment other energy sources, such as solar and wind energy. Besides, hydropower generation is better suited to meet demands for peak loads compared to thermal units.

These are some other advantages of hydropower generation

- Hydropower has furnished electricity to the world for more than a century, making it a time-tested, reliable technology of known costs and benefits.
- This is the most important renewable source of energy in the world. The Sun is the prime mover of the water cycle, and power can be produced as long as the rain/snow falls and rivers flow.
- As the raw material is virtually free, generating cost is almost free from inflation.
- Hydropower plants do not require much outlay on account of operation and maintenance and have long lives.
- Hydropower generation does not pollute the environment; no heat is produced, and no harmful gases are released.
- Hydropower power plants work at very high efficiency compared to the other major source (thermal power).
- Reservoirs can store water during times of low demand and can quickly start power generation. Thermal power plants take much longer to start up than hydropower plants.
- In multipurpose projects, generation of hydroelectric energy is combined with other uses such as irrigation, water supply, and flood control, resulting in tremendous savings of money and other resources.
- Small, mini-, and microscale hydropower projects can be tailor-made to provide power and minimize adverse environmental and social impacts.

- As a result of developments in turbine technology, efficient turbines whose capacities vary from several hundreds of MW to a few MW have been developed.

If waterpower is used to generate electricity in place of fossil fuels, emission of greenhouse gas can be substantially reduced. However, the creation of large reservoirs and accompanying infrastructures is being resisted by many local communities. Often, small hydropower projects built with the involvement of local communities provide an acceptable solution to electricity problems and the least harm to the environment. Many regions, particularly Asia, Europe, and North America, have developed substantial hydroelectric resources using small plants of <10 MW capacity.

DISADVANTAGES AND ISSUES IN HYDROPOWER GENERATION

The development of hydropower in many countries is not at desired rates for several reasons. Major hydropower projects have large initial costs and long gestation periods. Of late, these projects are being opposed because of harm that they can do to the environment. The reservoirs inundate forests, disturb wild life, and the local population may be displaced causing social tensions.

Careful planning and operation of hydropower facilities can minimize environmental damage, but environmental costs may hinder the development of hydropower in some areas. Another problem is the adverse effect of dams on river ecosystems. After construction of a dam, the river flow downstream is considerably reduced leading to significant changes in river ecology. The construction of a dam also restricts fish movement.

Proponents of these projects argue that many of the adverse impacts are temporary: they arise mainly during the construction phase. Problems such as forest submergence can be tackled by compensatory afforestation. The acceleration in regional development as a result of these projects can more than offset the harmful effects. Most problems are not insurmountable and can be largely overcome by careful planning and involving all stakeholders in decision-making.

COMPONENTS OF HYDROPOWER PROJECTS

In a storage project, the reservoir behind the dam stores water that generates electric power. The portion of the reservoir that is immediately upstream of the intake structure is known as the forebay. A penstock is a conduit to carry water from the forebay to turbines and gates, and valves are installed to control the flow of water. A surge tank is constructed to handle the problems of water hammer. The turbines and generators are installed in a powerhouse. The water comes out of the turbines through the draft tube and joins the tail water.

The term "static head" (see Fig. 1) denotes the difference between the water surface elevation in the forebay and the tail water level (TWL). The 'net head'

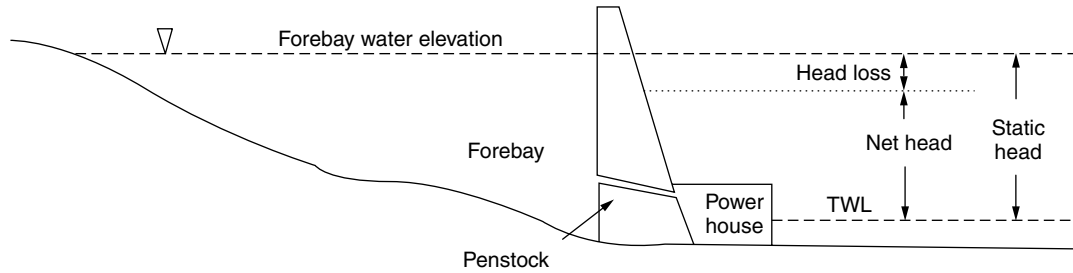


Figure 1. Terms related to head in a hydropower plant.

is the static head less losses in the penstock:

$$\begin{aligned} \text{Net head} &= \text{Static head} - \text{Losses} \\ &= \text{Water surface elevation in forebay} \\ &\quad - \text{TWL-Penstock losses} \end{aligned} \quad (1)$$

Although the head is related to the height of the dam, a low dam can yield a high head if the powerhouse is located some distance downstream from the dam. The amount of hydropower generated is a function of the discharge and the hydraulic head:

$$P = 9.817QH\eta \quad (2)$$

where P is electric power in kW, Q is the discharge through the power plant in m^3/s , and H is the net head in m. Further, η is the overall efficiency of the power plant expressed as a ratio (usually about 0.85) obtained by multiplying the turbine efficiency by the generator efficiency. Note that hydroelectric power generation depends on the rate of flow through the turbines and the effective head. Thus, power output can be controlled by releasing more water at a low head or less water at a high head. Most hydropower plants are constructed in hilly areas where steep slopes provide high heads.

TYPES OF HYDROPOWER PROJECTS

There are four major classifications of hydroelectric projects with respect to types of site development: storage, barrages, run-of-river, and pumped storage. Storage plants usually have heads in the medium to high range (greater than 25 m) and can store relatively large volumes of water during periods of high stream flow to provide supply for power generation during periods of low stream flow. The power plant is commonly located at the toe of the dam, though it might also be away from the dam. Peaking operation is frequently associated with storage projects, and this requires large and sometimes rapid fluctuations in releases of water. It is often necessary to provide facilities to even out the fluctuations in the discharge, if rapid changes in water levels below the project are not tolerable.

A barrage, also known as pondage, has a very small storage capacity. It can't effectively regulate flow and generates power according to the variation of load. Run-of-river plants have little or no storage and, therefore, must generate power from stream flow as it occurs with

little or no benefit from at-site regulation. Run-of-river projects generally have productive heads in the low to medium range (5 to 30 m) and are frequently associated with navigation or other multipurpose developments. For a base-load run-of-river project to be feasible, the stream must have a relatively high baseflow. Falls in irrigation canals are sometimes used to generate energy.

PUMPED STORAGE SCHEMES

A significant and growing portion of the hydroelectric capacity worldwide is devoted to pumped storage facilities that are designed solely to provide power during peak loads. These projects depend on pumped water as a partial or total source of water. Pumped storage projects consist of a high level forebay where inflow or pumped water is stored until needed and a low level afterbay where the power releases are stored. Pumping and generation are done by units composed of reversible pump turbines and generator motors connecting the forebay and afterbay. During off-peak hours, excess electricity produced by conventional power plants is used to pump water from lower to higher level reservoirs. During periods of high demand, water is released from the upper reservoir to generate electricity. Such projects derive their usefulness because the demand for power is generally low at night and on weekends, and therefore, pumping energy at a very low cost will be available from idle generating facilities. The feasibility of pumped storage developments arises from the need for relatively large amounts of peaking capacity, the availability of pumping energy at a cheap rate, and a sufficiently long off-peak period. It results in large cost savings through more efficient use of base-load plants.

There are three types of pumped storage development: diversion, off-channel, and in-channel. In the diversion type of development, water is released through generating units into an afterbay in an adjacent basin. The off-channel type of pumped storage development is most suitable when a forebay site exists on a hill above a stream where an afterbay can be constructed. The head differential should be large, and the forebay site should be close to the afterbay to avoid head loss and reduce construction costs. The water requirement in this scheme is not large after the initial supply has been provided.

In the in-channel type of scheme, the reservoir of a conventional power project is used as a forebay. The afterbay could be a reservoir from a downstream project or a reservoir provided solely to serve as an afterbay. This

scheme is more attractive if the cost of the afterbay is shared with other purposes.

LOAD OF HYDROPOWER PROJECTS

The demand for electrical energy is known as load. The ratio of the average power demand to peak power demand for the time period under consideration is known as the load factor, and it is computed daily, weekly, monthly, or annually. Thus,

$$\text{Load factor} = \frac{\text{Average power demand}}{\text{Peak power demand}} \quad (3)$$

With respect to the type of load served, hydropower projects can be classified in two categories: base-load plants and peaking plants. Base-load plants are projects that generate hydroelectric power to meet the base-load demand (the demand that exists 100% of the time).

Peaking plants generate power to supplement base-load generation during periods of peak demands. These plants must have sufficient capacity to satisfy peak demands, and enough water should be available. In general, a peaking hydroelectric plant is desirable in a system that has thermal generation units to meet base-load demands.

ESTIMATING HYDROPOWER POTENTIAL

Hydroelectric power potential is determined on the basis of the critical period, as indicated by the historical stream flow record. If a project serves more than one purpose and if, while serving another purpose, some of the storage or stream flow is not available for power production, the stream flow data should be adjusted to reflect the "loss." Losses such as evaporation and leakage must also be deducted from the available flow. The amount of power generated over a time, or energy, is expressed in kilowatt-hour (kWh):

$$\text{kWh} = 9.817QHT\eta \quad (4)$$

in which kWh is the hydropower generated during the period (kWh) and T is the number of hours in the period. Two methods are used to estimate the hydropower potential at a given site: the flow duration curve method and the sequential stream flow routing (SSR) method. In the first method, the flow duration curve developed at the site using the stream flow data is the basic input. The net head for various discharges is estimated, and the hydropower equation is used to estimate the power generated at many points on the flow duration curve. In this way, a power duration curve is developed. The

average annual energy and dependable capacity can now be calculated. This method is simple and fast, but it cannot take into account the installed capacity and key project features, such as power plant characteristics.

In the SSR method, the time step size and period of analysis are chosen, and the operation of the reservoir is simulated. For each time period, the reservoir outflow is computed, and the amount of energy generated is calculated using Equation 4. The process is repeated for all the time steps. Now the average annual, monthly, etc., generation as well as firm power can be computed. This method can take into account the reservoir and power plant characteristics. The results of this method are more realistic compared to the flow duration curve method.

OPERATION OF HYDROPOWER RESERVOIRS

Hydropower reservoirs store water to ensure supply as well as hydraulic head. During the filling season, the aim usually is to end the season with as much energy (as water) as possible in store in the system. The water stored in an upper reservoir (higher elevation) has higher potential energy, so upper reservoirs in a series should be filled first. After generating energy at an upper reservoir, the water can be captured in a downstream (lower) reservoir where energy can be generated again. The same logic also holds for any spill from an upper reservoir. Of course while storing water, one also has to examine the compatibility of hydropower use with the other uses of water.

During the drawdown, the objective is to maximize hydropower production for a given total storage amount vis-à-vis the demands. Recall that a system can generate the maximum amount of power when all reservoirs are full (the hydraulic heads will be the highest). If the available water is limited, it should be allocated among the reservoirs to maximize hydropower production. The governing variables are the storage and power generating capacities of reservoirs, the inflows, and the efficiencies of power plants. Note that in smaller reservoirs, the rate of increase of head per unit volume of additional water is higher compared to a large reservoir (that has more surface area). Referring to Fig. 2, the volume of water needed to increase the head by x units in a smaller reservoir (V_1) is less than the volume needed (V_2) to increase the head in a larger reservoir by the same increment.

The combination of the important variables in hydropower production, reservoir capacity, volume of inflows, and efficiency of power plant determines the ranking of the reservoirs to produce power. When reductions in storage are necessary, they are made from reservoirs that are least able to produce power. Conversely, an increase

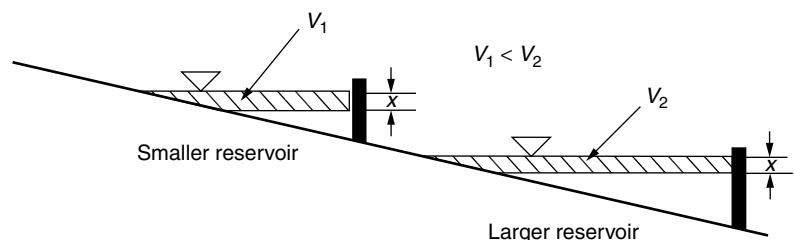


Figure 2. Change in head as capacities vary (after Reference 2).

in storage should be in reservoirs that have the greatest ability to produce power. The following reservoir ranking ratio can be used (2):

$$V_i = a_i e_i \left(\sum_{j=1}^i I_j \right) \quad (5)$$

where V_i is the increased power production per unit increase in storage, a_i is unit change in hydropower head per unit change in storage, and e_i is the power generation efficiency, all for reservoir i , and I_j is direct inflows and releases into reservoir j , for all reservoirs upstream of reservoir i . Note that reservoir 1 is the uppermost reservoir in the series. The reservoirs are ordered according to V_i values, and filling begins at the highest value of V_i and proceeds in descending order.

A number of software packages are available to analyze reservoir operation for hydropower generation. See HEC (3), McMahon and Mein (4), and Wurbs (5) for further details.

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HYDROELECTRIC RESERVOIRS AS ANTHROPOGENIC SOURCES OF GREENHOUSE GASES

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OVERVIEW

The ever-increasing demand for energy over the recent development of societies has spurred the construction

of hydroelectric facilities. Since dams were first used to generate hydropower around 1890, their construction rate increased tremendously to peak during the 1950s and the 1980s (1). Today, about 25% of the 33,105 large dams (≥ 15 m height) listed by the International Commission on Large Dams (ICOLD) are used for hydropower generation (2) and currently provide 19% of the world's electricity supply (1). Although over 150 countries operate hydroelectric plants, Brazil, China, Canada, Russia, and the United States produce more than 50% of the world's hydropower (1). According to data from 1996, hydroelectric reservoirs worldwide cover an estimated 600,000 km² (3).

Apart from the benefits they provide, hydroelectric reservoirs entail several social and environmental drawbacks such as the loss of lands and displacement of peoples (1,4), downstream hydrological alterations (5–8), elevated methylmercury burdens in fish and human consumers (8–11), as well as the loss of biodiversity (1,8). Recently, they also proved to be nonnegligible sources of greenhouse gases (GHG) like carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) (12–31). Owing to their implication in climate change (32–35), hydroelectric reservoirs have raised a sustained interest among both the scientific and political communities. Research during the last decade has brought valuable insights into the biogeochemical impacts stemming from their creation. Although it is now recognized that impoundment leads to both the disappearance of a GHG sink and the creation of a new anthropogenic GHG source (8,34), the duration of anthropogenic GHG emissions from these water bodies remains a matter of current debate (36).

ESTIMATES OF GHG EMISSIONS FROM HYDROELECTRIC RESERVOIRS WORLDWIDE

Table 1 provides statistical parameters for GHG fluxes at the surface of hydroelectric reservoirs. Spatial variability is an important issue that may interfere with regional estimates of GHG fluxes (31,33,35,36). Nevertheless, several data sets shown in Table 1 are normally distributed and mean values can therefore be used to estimate GHG fluxes on a latitudinal basis.

Exceptions are tropical CH₄ diffusive fluxes, which are not normally distributed, and N₂O diffusive fluxes for which data are too scarce. Likewise, bubbling fluxes are marred by either failure to comply with normality or insufficiency of data. In all of these cases, mean values must be used with caution as they poorly reflect the actual situation.

Generally speaking, tropical hydroelectric reservoirs possess the highest mean GHG fluxes, whereas the lowest mean fluxes are observed in temperate ones. A striking characteristic is the presence of CO₂ influxes in some temperate hydroelectric reservoirs from arid and semiarid regions (31). These distinctive GHG emission regimes are caused mainly by particular environmental conditions, such as temperature, which can influence GHG production processes (22,33,36). To a lesser extent, important differences in the mean age of hydroelectric reservoir sets in Table 1 may also intervene (33,35,36).

Table 1. GHG Fluxes at the Air/Water Interface of Hydroelectric Reservoirs Worldwide During the Open Water Period (12,14–29,31,37,38)

		Latitude (Climate) (Number of Reservoirs) ^a	Mean Age (y)	Range (mg·m ⁻² ·d ⁻¹)	Median (mg·m ⁻² ·d ⁻¹)	Mean (mg·m ⁻² ·d ⁻¹)	Std Dev. (± mg·m ⁻² ·d ⁻¹)	Normality
Diffusive Fluxes	<i>f</i> CO ₂	Boreal (12)	24	653 to 2500	1346	1459	600	yes
		Temperate (16)	44	−1195 to 2200	685	525	938	yes
		Tropical (21)	13	−142 to 13737	4789	5467	3746	yes
	<i>f</i> CH ₄	Boreal (7)	29	3.5 to 22.8	10.0	10.8	6.5	yes
		Temperate (13)	49	1.3 to 15.0	7.0	6.7	3.7	yes
		Tropical (22)	16	5.7 to 233.3	21.7	50.5	66.0	no
	<i>f</i> N ₂ O	Boreal (3)	18	0.02 to 0.5	0.2	0.2	0.2	yes
		Temperate (0)	ND	ND	ND	ND	ND	ND
		Tropical (4)	8	5.6 to 800	34.8	218.8	388.3	no
Bubbling Fluxes	<i>f</i> CO ₂	Boreal (2)	10	1.0	1.0	1.0	0	ND
		Temperate (1)	70	1.0	ND	ND	ND	ND
		Tropical (16)	14	0.02 to 26	0.3	2.5	6.5	no
	<i>f</i> CH ₄	Boreal (5)	20	0.04 to 184.2	0.8	46.4	79.6	no
		Temperate (1)	70	14	ND	ND	ND	ND
		Tropical (21)	15	0 to 800	22.9	85.6	179.1	no
	<i>f</i> N ₂ O	Boreal (2)	27	0 to 0.03	0.02	0.02	0.02	ND
		Temperate (0)	ND	ND	ND	ND	ND	ND
		Tropical (0)	ND	ND	ND	ND	ND	ND

^aHydroelectric reservoirs that have been sampled multiple times are considered as separate water bodies.

GHG DYNAMICS IN HYDROELECTRIC RESERVOIRS

GHG emissions are common in natural freshwater ecosystems (39–42). However, hydroelectric reservoirs differ from these natural ecosystems in several fundamental aspects (43–45) and therefore cannot be directly compared. Typically, these man-made water bodies are created by damming rivers or lakes. Consequently, the first outcome of hydroelectric reservoir creation is the flooding of terrestrial ecosystems and the subsequent death of the associated vegetation (10,13,33). Overall, this event likely corresponds to the permanent loss of a potential carbon sink (10,33,46), as exemplified by data in Table 2. The second outcome attributable to hydroelectric reservoirs is the creation of a net source of GHG through perturbations that are not observed in steady, natural aquatic ecosystems. These perturbations are highlighted by the GHG

emission regimes of hydroelectric reservoirs, which generally overwhelm, at least temporarily, those of natural water bodies (10,13,15,16,21–23,33,35,56).

Emissions that would not have existed without the perturbations brought about by hydroelectric reservoirs are referred to as “anthropogenic GHG emissions.” Consequently, when evaluating these emissions, the following issues must be taken into account. First, the establishment of a valid reference state must be based on a comparison with GHG fluxes prior to flooding because of the major impacts of impoundment on terrestrial ecosystems (13,36). Second, the altered GHG dynamics following the impoundment phase as well as its temporal evolution must be considered (36).

Figure 1 summarizes the current knowledge on GHG dynamics in hydroelectric reservoirs. GHG emissions result from the superposition of short- and long-term

Table 2. Examples of Preimpoundment Carbon Fluxes from Terrestrial Ecosystems Commonly Flooded for the Creation of Hydroelectric Reservoirs^a

Terrestrial Ecosystem	Study Site	Flux (g C·m ⁻² ·y ⁻¹) ^b	Reference
Boreal mesotrophic lake	Canada	+73	10
Boreal peatland (<i>Sphagnum</i> spp., <i>Ledum groenlandicum</i>)	Canada	–20 to –50	10, 47
Boreal aspen (<i>Populus tremuloides</i>) forest, mature	Canada	–80 to –290	48
Boreal pine (<i>Pinus sylvestris</i>) forest, mature	Finland	–214 to –252	49
Boreal spruce (<i>Picea mariana</i>) forest, mature	Canada	–60 to –158	48, 50
Temperate beech (<i>Fagus grandifolia</i>) forest, mature, CO ₂ only	Germany	–490 to –494	51
Temperate pine (<i>Pinus taeda</i>) forest, CO ₂ only	United States	–69	52
Temperate Mediterranean alpine grassland, CO ₂ only	United States	+29 to –132	53
Tropical mesic savanna, considering/not considering bush fires	Australia	–100/–300	54
Tropical rain forest, mature	Brazil	+80 to –390	55
Natural lakes, worldwide average, CO ₂ only	many countries	+70	40

^aCarbon storage capacity of terrestrial ecosystems depends on several factors, such as their type, maturity level, and disturbance history, as well as climatic conditions (9,14,36,49–51).

^bPositive sign indicates a carbon source; negative sign indicates a carbon sink.

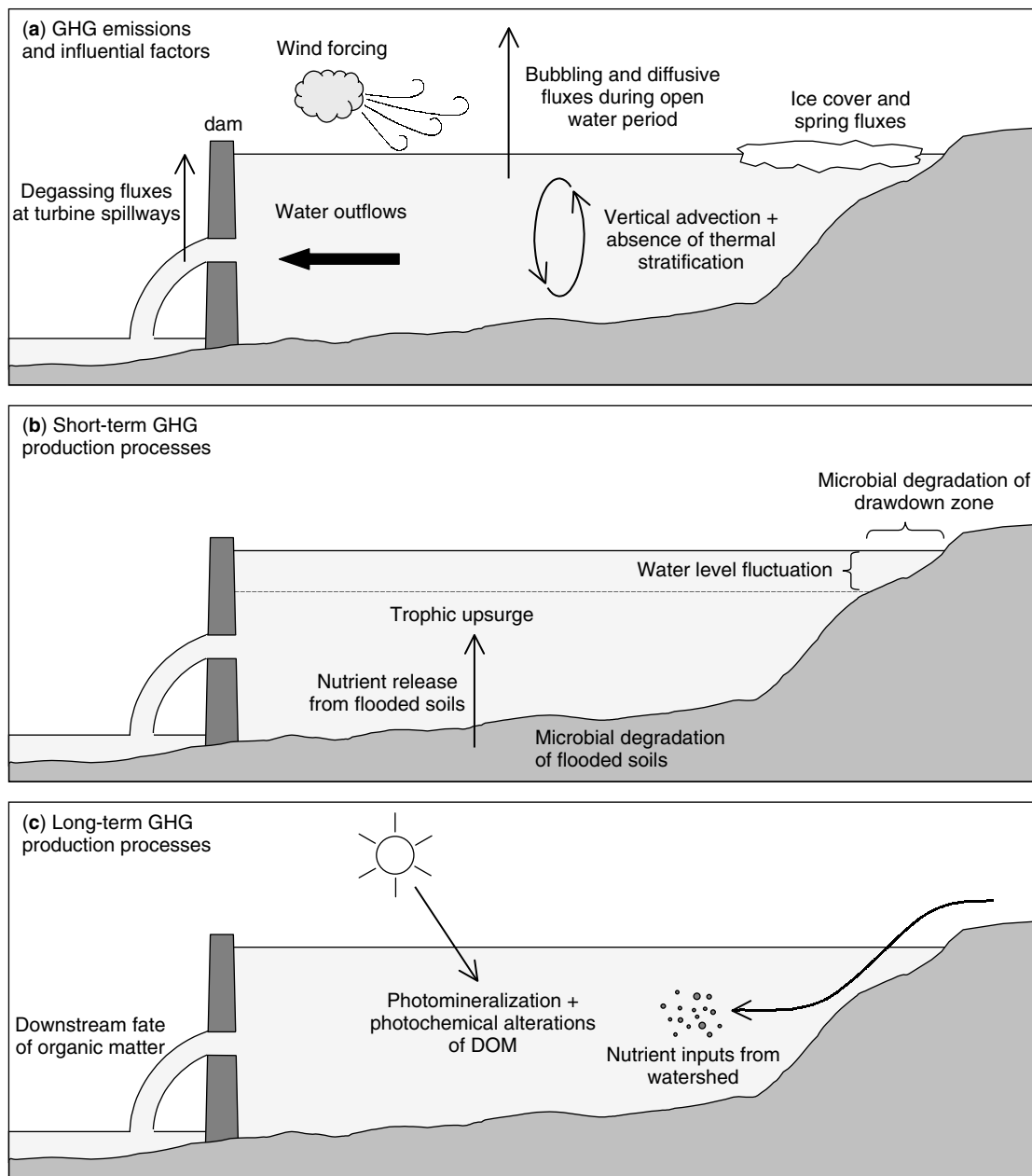


Figure 1. Dynamics of GHG in hydroelectric reservoirs. GHG emissions (a) are the result of short-term (b) and long-term (c) production processes.

production processes. Fundamental mechanisms involved in the production and emission of GHG are exhaustively described in limnology handbooks (57,58). Therefore, only the characteristics of hydroelectric reservoirs that lead to anthropogenic GHG emissions are hereafter described.

GHG Emission Processes and Influential Factors in Hydroelectric Reservoirs

GHG Fluxes at the Air/Water Interface During the Open Water Period. During the open water period, a fraction of GHG escapes from hydroelectric reservoirs through vertical advection (bubbling fluxes) (59) and molecular diffusion (diffusive fluxes) (60,61). GHG emission levels

at the air/water interface may vary widely both within and between hydroelectric reservoirs, depending on the factors presented in Table 3.

Hydrometeorological forcings induce a particular hydrodynamic within hydroelectric reservoirs that enhance air/water interface GHG fluxes (36,43). For instance, wind generates surface waves and the vertical advection (Langmuir circulation) of water masses (43), thereby promoting gas transfer velocity across the air/water interface (61,62). For hydroelectric reservoirs with relatively extended surface areas and large fetches, wind leverage can be substantial (14,22,63,64). The operation regime of hydroelectric reservoirs also contributes to hydrodynamic features, which augment

Table 3. Main Factors Identified to Be Responsible for Inter- and Intra-Reservoir Variations in GHG Emission Levels at the Air/Water Interface (16,21,22,27,31,36)

Inter-Reservoir Variability	
Internal factors(pertaining to hydroelectric reservoir)	age of impoundment surface area morphometry and bathymetry physicochemistry operation regime
Regional factors(pertaining to watershed)	type of ecosystems flooded climatic feature pedology and geological basement ecological features
Intra-Reservoir Variability	
Spatial heterogeneities	water column depth water column temperature presence or absence of macrophytes hydrodynamics (main channel vs. protected coves) exposure to meteorological forcings (wind and sun)
Temporal heterogeneities	interannual (age, variations in climate and operation regime) season cycle (water level, biology, ice cover if applicable) day-night cycle (biology, especially photosynthesis)

GHG emissions. Intermittent huge water outflows for hydropower generation enable vertical advection and impede thermal stratification of the water column (43). This frequent situation in hydroelectric reservoirs (23,26,31,45) allows hypolimnetic water to mix with the upper layers, thereby promoting GHG evasion to the atmosphere.

GHG Fluxes at the Air/Water Interface During Spring. During wintertime, ice covers boreal and some temperate freshwater bodies. As GHG are still being produced throughout this period (26), gas bubbles accumulate under the ice. When the ice melts in the spring, GHG evade to the atmosphere at greater rates than during the open water period (22,26). However, as most studies on hydroelectric reservoirs are conducted in summer, spring GHG fluxes are often overlooked.

Degassing Fluxes at Turbine Spillways. GHG also escape from the water column at turbine spillways when hydropower is produced. Such degassing fluxes are caused by the sudden change in pressure as well as the increased air/water exchange surface (17,19) and strongly depend on the volume of outflows, the GHG concentrations within the water column, and the depth of water outlets on the dam. Although an important term of the GHG budget of hydroelectric reservoirs (36), degassing fluxes have seldom been assessed (19,31,46).

GHG Production Processes in Hydroelectric Reservoirs

Flooded Soils and Drawdown Zones. Hydroelectric reservoirs create a new interface between water and soils, thereby triggering the onset of major biogeochemical perturbations. Following impoundment, the physicochemical features of flooded soils promote the production

of GHG through microbial degradation of organic matter (10,13). Biomass degradation is generally associated with oxygen depletion at the bottom of tropical hydroelectric reservoirs (19,23,28,65), a phenomenon that seems to be transient or inexistent in boreal and temperate ones (16,26,31,66).

Drawdown zones refer to the extended portion of hydroelectric reservoir banks that are periodically flooded by fluctuating water levels that occur during the operation phase. These zones act as intermittent water/soil interfaces and also constitute favorable sites for the production of GHG (67).

Typically, flooded organic matter undergoes an initial phase of rapid decomposition of labile compounds, followed by the slower decomposition of more recalcitrant compounds (13,28). Microbial degradation of flooded soils thus decreases with time and, according to studies on both carbon loss and isotopic tracers ($\delta^{13}\text{C}$), appears to be limited to approximately ten years (68,69).

Trophic Upsurge. Following impoundment, a fraction of compounds from flooded soils and drawdown zones are released into the water column of hydroelectric reservoirs (28,68,70). This nutrient (C, N, P) pulse increases water turbidity, alters feeding habits as well as trophic levels of the planktonic community, and ultimately disrupts its initial photosynthesis:respiration ratio (22,69–73). Overall, these drastic shifts within the planktonic community structure provoke the transition of the aquatic system to a more allotrophic state (trophic upsurge) (74). Consequently, respiration overwhelms photosynthesis, thus enhancing the production of GHG in the water column.

Figure 2 illustrates the temporal progression of bacterioplankton in hydroelectric reservoirs. After the impoundment phase, bacterial biomass increases sharply (Fig. 2a). Then, as nutrient inputs from flooded soils gradually decrease, physicochemical parameters within the

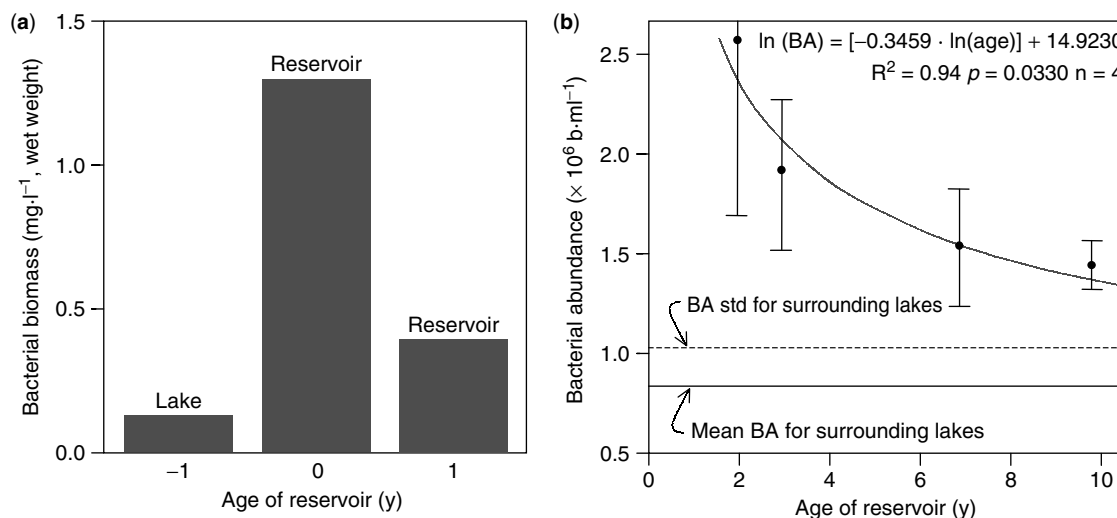


Figure 2. Temporal progression of bacterioplankton in reservoirs. Year “0” corresponds to the impoundment event. (a) Bacterioplankton biomass before and after the impoundment of a experimental reservoir created from a small boreal lake (70). (b) Evolution of bacterioplankton abundance (BA) following the impoundment of the boreal hydroelectric reservoir Laforge-1 (22,38,75). Error bars are standard deviation.

water column tend to evolve toward natural conditions (66,76,77). In accordance with the bottom-up theory, bacterial abundance also declines (65,70), but, as suggested in (Fig. 2b), a substantial time lag exists between the depletion of flooded nutrients and the anticipated onset of an equilibrium state within the bacterioplanktonic population.

Allochthonous Inputs from the Watershed. The actual contribution of allochthonous organic matter to anthropogenic GHG emissions from hydroelectric reservoirs is unclear (36). Nevertheless, the fact that long-term GHG emissions from certain hydroelectric reservoirs still overwhelm those of natural water bodies (33,35) and do not stem from the degradation of flooded soils (68,69) supports the hypothesis that these anthropogenic emissions are fueled by inputs from the watershed (22,36,56,78). If so, allochthonous inputs would only contribute to anthropogenic GHG emissions from hydroelectric reservoirs if they are mineralized to a greater extent than in natural ecosystems.

Two phenomena may account for such a case. On the one hand, photochemical reactions play a key role in the carbon cycle of aquatic systems (79–84) through both direct photomineralization of dissolved organic matter (DOM) into CO₂ (85–89) and molecular alterations that modify the bioavailability of DOM to bacteria (90,91). Empirical data suggest that these mechanisms play a greater role in hydroelectric reservoirs than in natural water bodies. For instance, a study conducted on a boreal hydroelectric reservoir has indicated that DOC photoreactivity exceeded that of boreal and temperate natural lakes significantly (94). This situation is not due to qualitative aspects of DOM, but rather to differences in age of water, as well as iron and manganese concentrations (92).

On the other hand, hydroelectric reservoirs disrupt natural streamflows and increase the water residence time (7,8,36,93), thereby retaining organic matter and hypothetically allowing for increased degradation. It is thus likely that hydroelectric reservoirs modify the mineralization pattern of organic matter at the scale of the watershed. However, this important issue has yet to be addressed.

DURATION OF ANTHROPOGENIC GHG EMISSIONS FROM HYDROELECTRIC RESERVOIRS

As shown in (Fig. 3) and elsewhere (14,33,35), GHG emissions significantly decline with the age of hydroelectric reservoirs. The duration of anthropogenic GHG emissions is closely related to the perturbations caused by their creation and operation. As these perturbations occur at different time scales (Fig. 1), anthropogenic GHG emissions from hydroelectric reservoirs may last for decades (33,35).

The duration of anthropogenic GHG emissions from hydroelectric reservoirs can be grossly estimated through a heuristic comparison with natural lakes. Higher GHG emission levels from the former likely indicate that certain biogeochemical perturbations still intervene. (Fig. 3) suggests that CO₂ diffusive fluxes exceeding the mean of natural lakes worldwide (704 mg CO₂ · m⁻² · d⁻¹) (40) can last up to 70 years. However, weak regression coefficients (R²) between the magnitude of GHG fluxes and the age of hydroelectric reservoirs, in (Fig. 3) and in previous literature reviews (33,35), indicate that the latter parameter poorly explains the decline of GHG fluxes. Ultimately, the accurate determination of the duration of anthropogenic GHG emissions will depend on an exhaustive appraisal of the mechanisms involved in the production of GHG.

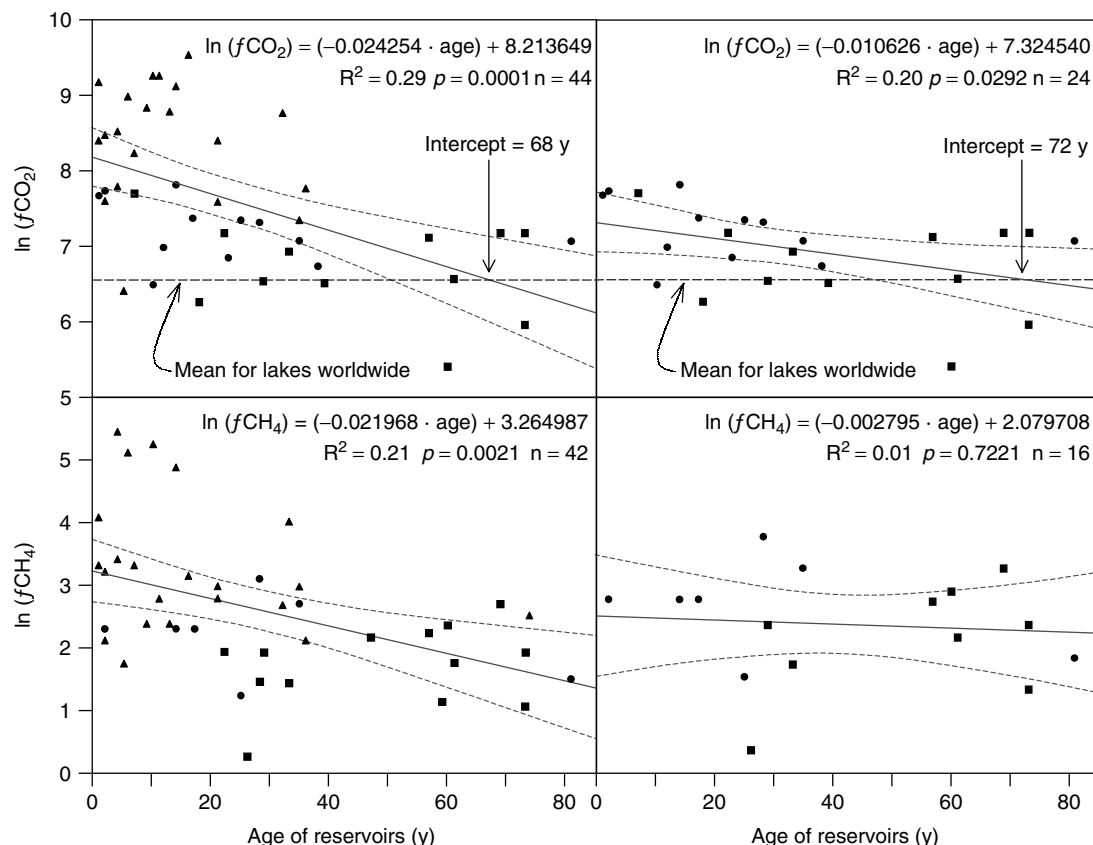


Figure 3. Temporal evolution of CO₂ and CH₄ diffusive fluxes from hydroelectric reservoirs during the open water period (12,14–16,19–29,31,37,92). Left panel: Includes hydroelectric reservoirs from all climatic zones (boreal, temperate, and tropical). Right panel: Boreal and temperate hydroelectric reservoirs only. Hydroelectric reservoirs with CO₂ influxes have been discarded and those that have been sampled during different years are considered as separate water bodies. Circles = boreal; squares = temperate; triangles = tropical. Dashed curved lines represent the 95% confidence intervals.

POLITICAL ASPECTS

As a result of the threat of climate change, anthropogenic GHG emissions have become of major concern for political entities. In 1990, the Intergovernmental Panel on Climate Change's (IPCC) First Assessment Report set the basis for the United Nations Framework Convention on Climate Change (UNFCCC), which was further implemented by the Kyoto Protocol in 1997 (34). The UNFCCC establishes an overall international structure to deal with climate change. Its mandate is to require signatory countries to gather relevant information on GHG emissions (national GHG inventories) as well as to develop strategies to prevent or mitigate these emissions (94).

From this perspective, the energy sector represents an area where GHG abatements are possible through strategies such as the "clean development mechanism" (CDM). As defined by the Marrakesh Accords, a project is considered a CDM provided that it substitutes a higher GHG-emitting option (95) [for instance, building hydropower plants instead of thermal ones, the latter being commonly recognized as emitting more GHG (28,33,35)]. Certificates of GHG emission reduction can then be acquired through the World Bank's Prototype Carbon Fund (PCF) (96).

Within the framework of the CDM, anthropogenic GHG emissions from hydroelectric reservoirs cannot be ignored. However, their inclusion into national GHG inventories is a prerogative that has yet to be achieved. To date, only GHG emissions produced by the degradation of flooded soils are considered as anthropogenic by IPCC (97). Nevertheless, the eventual part of GHG emissions stemming from allochthonous inputs should also be considered in the next IPCC guidelines, provided that enough evidence exists to substantiate such a situation. In any case, the inclusion of hydroelectric reservoirs into national GHG inventories is planned for the second commitment under the Kyoto Protocol, effective as of 2012. Countries that are characterized by large surface areas covered by hydroelectric reservoirs should then observe a significant increase in their national GHG inventories.

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HYDROLOGIC PERSISTENCE AND THE HURST PHENOMENON

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INTRODUCTION

Unlike common random series like those observed, for example, in games of chance (dice, roulette, etc.), hydrologic (and other geophysical) time series have some structure, that is, consecutive values of hydrologic time series depend on each other. It is easy to understand that, for instance, in a monthly river flow series, a month of very high flow is likely to be followed by a month of high flow, too. River flow is interrelated to groundwater storage, so high flow indicates that groundwater storage will be high, too, and given that the groundwater flow is a slow process, it is expected that its contribution to river flow in the next month can be significant. This explains the dependence of consecutive values of hydrologic time series, which have been known as *short-range* (or *short-term*) *dependence, persistence, or memory*.

Interestingly, however, there is another kind of dependence observed on larger timescales, known as *long-range* (or *long-term*) *dependence, persistence, or memory*. This was discovered by Hurst (1), while investigating the discharge time series of the Nile River in the framework of the design of the Aswan High Dam, and was found in many other hydrologic and geophysical

time series. This behavior is the tendency of wet years to cluster into multiyear wet periods or of dry years to cluster into multiyear drought periods. The terms “Hurst phenomenon” and “Joseph effect”, due to Mandelbrot (2) from the biblical story of the “seven years of great abundance” and the “seven years of famine,” have been used as alternative names for the same behavior. Since its original discovery, the Hurst phenomenon has been verified in several environmental quantities such as (to mention a few of the more recent studies) in wind power (3); global or point mean temperatures (4–6); flows of several rivers such as the Nile (7,8); the Warta, Poland (9), Boeotikos Kephisos, Greece (5), and the Nemunas, Lithuania (10); inflows of Lake Maggiore, Italy (11); indexes of North Atlantic oscillation (12); and tree-ring widths, which are indicators of past climate (8). In addition, the Hurst phenomenon has gained new interest today due to its relation to climate changes (5,13–15).

The possible explanation of long-term persistence must be different from that of short-term persistence discussed above. This will be discussed later. However, its existence is easy to observe even in a time series plot, provided that the time series is long enough. For example, in Fig. 1 (up) we have plotted one of the most well-studied time series, that of the annual minimum water level of the Nile River for the years 622–1284 A.D. (663 observations), measured at the Roda Nilometer near Cairo (16, pp. 366–385; 17). In addition to the plot of the annual data values versus time, the 5-year and 25-year averages are also plotted versus time. For comparison, in the lower panel of Fig. 1, we have also plotted a series of white noise (consecutive independent identically distributed random variates) with

statistics the same as those of the Nilometer data series. We can observe that the fluctuations in the aggregated processes, especially of the 25-year average, are much greater in the real-world time series than in the white noise series. Thus, fluctuations in a time series on a large scale distinguish it from random noise.

STOCHASTIC REPRESENTATION OF THE HURST PHENOMENON

Quantification of long-term persistence is better expressed mathematically using the theory of stochastic processes. Let X_i denote a stochastic representation of a hydrometeorologic process where $i = 1, 2, \dots$, denotes discrete time with a time step or scale which for the purposes of this article is annual or multiannual. It is assumed that the process is stationary, a property that does not hinder exhibiting multiple scale variability. The stationarity assumption implies that its statistics are not functions of time. Therefore, we can denote its statistics without reference to time, that is, its mean as $\mu := E[X_i]$, its autocovariance as $\gamma_j := \text{Cov}[X_i, X_{i+j}]$ ($j = 0, \pm 1, \pm 2, \dots$), its autocorrelation $\rho_j := \text{Corr}[X_i, X_{i+j}] = \gamma_j/\gamma_0$, and its standard deviation $\sigma := \sqrt{\gamma_0}$. Further, we assume ergodicity, so that these statistics can be estimated from a unique time series substituting time averages for expected values.

Let k be a positive integer that represents a timescale larger than the basic timescale of the process X_i . The aggregated stochastic process on that timescale is denoted as

$$Z_i^{(k)} := \sum_{l=(i-1)k+1}^{ik} X_l \quad (1)$$

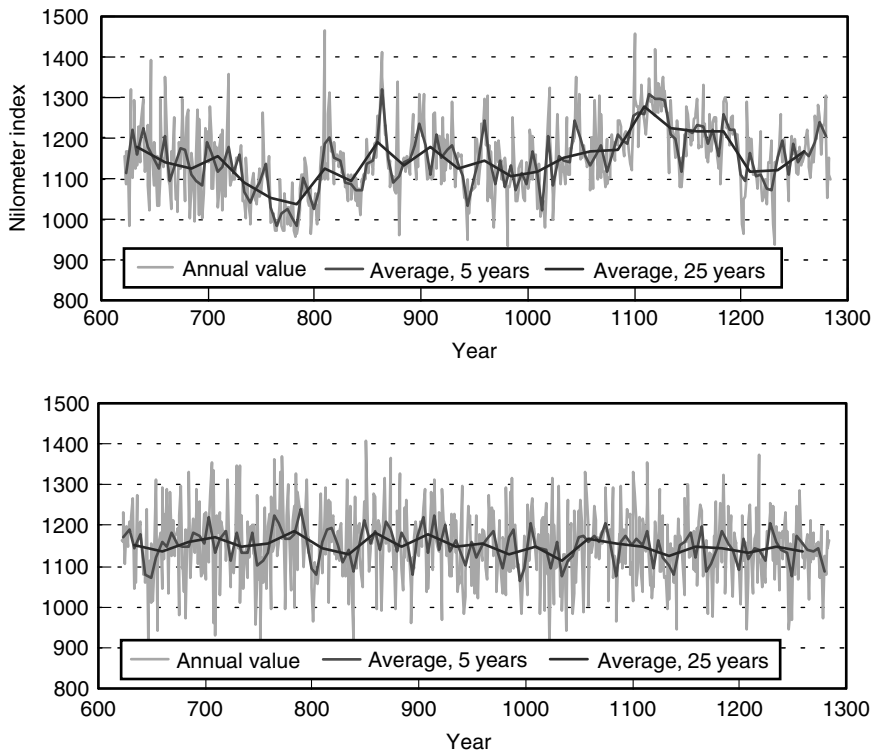


Figure 1. (Up) Plot of the Nilometer series indicating the annual minimum water level of the Nile River for the years 622–1284 A.D. (663 years); (down) a white noise series with same mean and standard deviation, for comparison.

The statistical characteristics of $Z_i^{(k)}$ for any timescale k can be derived from those of X_i . For example, the mean is

$$E[Z_i^{(k)}] = k \mu \quad (2)$$

and the variance and autocovariance (or autocorrelation) depend on the specific structure of γ_j (or ρ_j). In a process that exhibits the Hurst phenomenon, the variance $\gamma_0^{(k)}$ for timescale k is related to that of the basic scale γ_0 by

$$\gamma_0^{(k)} := \text{Var}[Z_i^{(k)}] = k^{2H} \gamma_0 \quad (3)$$

where H is a constant known as the Hurst coefficient whose values are in the interval $[0.5, 1]$. The value $H = 0.5$ corresponds to random noise, whereas values in the interval $[0, 0.5]$ are mathematically possible but without interest in hydrology. Consequently, the standard deviation is a power law of the scale or level of aggregation k with exponent H , that is,

$$\sigma^{(k)} := (\gamma_0^{(k)})^{1/2} = k^H \sigma \quad (4)$$

This simple power law can be easily used for detecting whether a time series exhibits the Hurst phenomenon and for determining the coefficient H , which is a measure of long-term persistence. Equation 4 calls for a double logarithmic plot of standard deviation $\sigma^{(k)}$ of the aggregated process $Z_i^{(k)}$ versus timescale k . In such a plot, called an aggregated standard deviation plot, Hurst behavior is manifested as a straight line arrangement of points corresponding to different timescales, whose slope is the Hurst coefficient. An example is depicted in Fig. 2 for the Nilometer series of Fig. 1. Clearly, the plot of the empirical estimates of standard deviation is almost a straight line on the logarithmic diagram of slope 0.85. For comparison, we have also plotted the theoretical curve for the white noise of slope equal to 0.5, significantly departing from historical data.

By virtue of Eq. 3, it can be shown that the autocorrelation function, for any aggregated timescale k ,

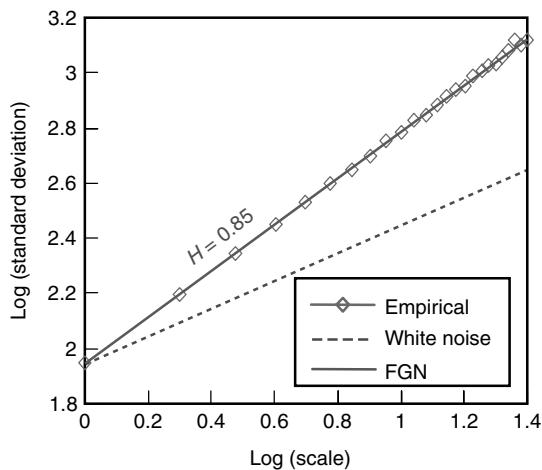


Figure 2. Aggregated standard deviation plot of the Nilometer series.

is independent of k and is given by

$$\begin{aligned} \rho_j^{(k)} &= \rho_j = (1/2)(|j+1|^{2H} + |j-1|^{2H}) - |j|^{2H} \\ &\approx H(2H-1)|j|^{2H-2} \end{aligned} \quad (5)$$

which shows that autocorrelation is a power function of the lag. Consequently, the autocovariance $\gamma_j^{(k)} = \gamma_0^{(k)} \rho_j^{(k)}$ is a power law of both the scale k (with exponent $2H$) and the lag j (with exponent $2H-2$).

The power spectrum of the process,

$$s_\gamma^{(k)}(\omega) := 2 \sum_{j=-\infty}^{\infty} \gamma_j^{(k)} \cos(2\pi j\omega) \quad (6)$$

is given approximately by

$$s_\gamma^{(k)}(\omega) \approx 4(1-H)\gamma_0^{(k)}(2\omega)^{1-2H} \quad (7)$$

which is a power law of both the scale k (with exponent $2H$) and the frequency ω (with exponent $1-2H$).

The power law Eqs. 5 and 7 can be used, in addition or alternatively to Eqs. 3 and 4, to detect the Hurst behavior of a time series. Note that Hurst's (1) original formulation to detect this behavior was based on another quantity, the so-called rescaled range, which corresponds to the cumulated process of inflow minus outflow of a hypothetical infinite reservoir.

Equations 3–7 describe the second-order properties of the process $Z_i^{(k)}$. A generalization is possible, if we assume that the process of interest exhibits scale invariant properties in its (finite-dimensional joint) distribution function:

$$(Z_i^{(k)} - k\mu) \stackrel{d}{=} \left(\frac{k}{l}\right)^H (Z_j^{(l)} - l\mu) \quad (8)$$

where the symbol $\stackrel{d}{=}$ stands for equality in distribution. In this case, Eq. 3 can be obtained from Eq. 8 by setting $i = j = l = 1$ and taking the variance of both sides. Equation 8 defines X_i and $Z_i^{(k)}$ as stationary increments of a self-similar process. If, in addition, X_i (and hence $Z_i^{(k)}$) follows the normal distribution, then X_i (and $Z_i^{(k)}$) is called fractional Gaussian noise (FGN; (18)). Our interest here includes processes that may be not Gaussian, so we will limit the scaling property of Eq. 8 to second-order properties only and call the related process a *simple scaling signal* (SSS).

PHYSICAL EXPLANATIONS OF THE HURST PHENOMENON

As described in the Introduction, the concept of short-term persistence in hydrologic processes is easy to explain, whereas long-term persistence and the Hurst phenomenon are more difficult to understand. Mesa and Poveda (19) classify the Hurst phenomenon as one of the most important unsolved problems in hydrology and state that “something quite dramatic must be happening from a physical point of view.” However, several explanations

have been proposed. These can be classified in two categories, physically based and conceptual.

Klemeš (20) proposed an explanation that may be classified in the first category. According to this, the Hurst behavior of hydrologic records can be explained by representing the hydrologic cycle by a “circular cascade of semi-infinite storage reservoirs” where the output from one reservoir constitutes an essential part of input into the next. He showed that, even with an originally uncorrelated Gaussian forcing, the outputs grew progressively more Hurst-like as the complexity of the system increased, for example, the number of reservoirs in the cascade. Another example of such a hydrologic system was suggested in Klemeš (21).

Beran (17, pp. 16–20) describes two physically based model types that lead to system evolution (in time or space) with long-range dependence. The first model type applies to critical phenomena in nature such as phase transition (transition from a liquid to a gaseous phase or spontaneous magnetization of ferromagnetic substances). For some critical system temperature, the correlation of the system state at any two points decays slowly to zero, so the correlation in space can be represented by Eq. 5. The second type is related to models based on stochastic partial differential equations, which, under certain conditions, result in solutions with long-range dependence. These models provide sound links of long-range dependence with physics but are very complex.

A simple model of this category was studied by Koutsoyiannis (6). This model assumes a system with purely deterministic dynamics in discrete time, which, however, results in a time series with an irregular appearance exhibiting the Hurst phenomenon. The system dynamics is based on the simple map

$$x_i = g(x_{i-1}; \alpha) := \frac{(2 - \alpha) \min(x_{i-1}, 1 - x_{i-1})}{1 - \alpha \min(x_{i-1}, 1 - x_{i-1})} \quad (9)$$

where x_i is the system state, assumed to be scalar, at time i and $\alpha < 2$ is a parameter. This map, known as a generalized tent map, has been used in the study of dynamic systems. For example, the map approximates the relation between successive maxima in the variable $x(t)$ from the Lorenz equations that describe climatic dynamics (22, p. 150). Koutsoyiannis (6) demonstrated that this model can describe a system subject to the combined action of a positive and a negative feedback. If the parameter α is assumed to vary in time following the same map, $\alpha_i = g(\alpha_{i-1}; \lambda)$, then we obtain the double tent map,

$$u_i = G(u_{i-1}, \alpha_{i-1}; \kappa, \lambda) := g(u_{i-1}; \kappa \alpha_{i-1}) \quad (10)$$

Both parameters κ and λ should be < 2 , whereas the domain of u_i is the interval $[0, 1]$. If the system domain is the entire line of real numbers, we can apply an additional transformation to shift from $[0, 1]$ to $[-\infty, \infty]$:

$$x_t = \ln[u_t/(1 - u_t)] \quad (11)$$

The model behavior with respect to parameters κ and λ is depicted in Fig. 3. We observe that for small values of

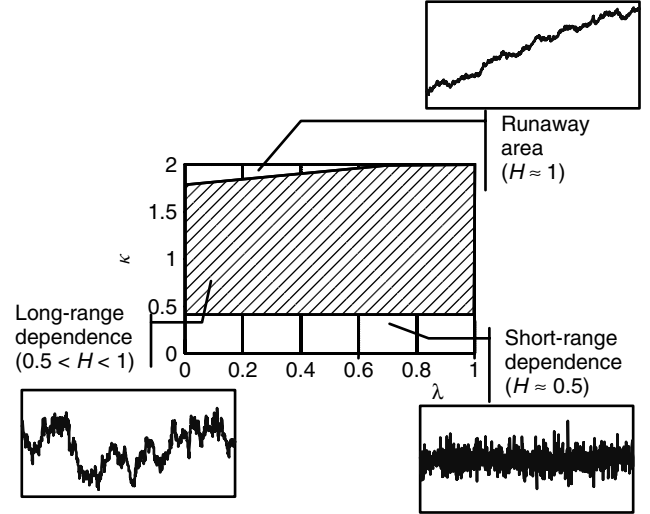


Figure 3. Schematic of the general behavior of the double tent map in terms of the ranges of its parameters κ and λ .

the parameter κ , the time series synthesized by the model exhibits short-range dependence with a Hurst coefficient around 0.5. For large values of κ , the model yields runaway behavior. However, in between the two noninteresting areas, there is an area of parameter values, shaded in Fig. 3, in which the resulting time series exhibits long-range dependence. All three types of behavior are observed for negative values of λ (not shown in Fig. 3), whereas for $\lambda > 1$, only the runaway behavior is observed, regardless of the value of κ .

The second category, labeled here “conceptual,” does not aim at explaining the physical mechanism leading to Hurst behavior of historical records of some natural or other processes but examines different stochastic mechanisms that might produce realizations resembling the patterns of the observed empirical time series. For example, Klemeš (20) analyzed several variants of the “changing mean” mechanism which assumes that the mean of the process is not a constant determined by the arithmetic mean of the record but varies through time. Specifically, he performed numerical experiments with the following Gaussian random processes:

1. a process with the mean alternating periodically between two values after constant time intervals called “epochs”;
2. a process with a monotonic linear trend in the mean, throughout the entire series length;
3. a process similar to 1 but with epoch lengths taking two different values with probabilities p and $1 - p$; and
4. a process with a Gaussian-distributed mean randomly varying from epoch to epoch, the epoch length also varying randomly and following either a uniform, exponential, or (single parameter) Pareto distribution.

The processes behaved increasingly Hurst-like as their structure changed from 1 to 4. This behavior was most

influenced by the distribution of epoch lengths, whereas the distribution of the mean itself had little effect.

The effect of periodical patterns, which are extensions of those of model 1, have been thoroughly studied by Montanari et al. (23), who, however, noted that such patterns are unusual in real data. The effect of monotonic deterministic trends, which are extensions of model 2, was studied by Bhattacharya et al. (24), who showed mathematically that a trend of the form $f(t) = c(m+t)^{H-1}$, where t denotes time, c a nonzero constant, m a positive constant, and H a constant in the interval $[0.5, 1]$, results in time series exhibiting the Hurst phenomenon with a Hurst coefficient precisely equal to H . We may note, however, that this kind of nonstationarity with a monotonic deterministic trend spanning the whole length of a time series can hardly represent a long time series of real data, even though in short time series it seems to be realistic. For example, to refer to the Nilometer series of Fig. 1, if one had available only the data of the period 700–800, one would detect a “deterministic” falling trend of the Nile level; similarly, one would detect a regular rising trend of the Nile level between the years 1000–1100. However, the complete picture of the series suggests that these trends are parts of large-scale random fluctuations rather than deterministic trends.

Based on this observation, Koutsoyiannis (8) proposed a conceptual explanation, which can be regarded as an extension of Klemes's model 4 and is also similar to other proposed conceptual models, as will be discussed later. More specifically, Koutsoyiannis (8) demonstrated that superimposition of three processes with short-term persistence results in a composite process that is practically indistinguishable from an SSS process.

This demonstration is reproduced here in Figs. 4–5. It starts by assuming a Markovian process U_i , like that graphically demonstrated in Fig. 4a, where mean $\mu := E[U_i]$, variance γ_0 , and lag one autocorrelation coefficient $\rho = 0.20$. The specific form of this process is an AR(1) one, $U_i := \rho U_{i-1} + E_i$, where E_i is white noise, and its autocorrelation is

$$\text{Corr}[U_i, U_{i+j}] = \rho^j \quad (12)$$

The autocorrelation function is shown in Fig. 5a along with the autocorrelation function of SSS with the same lag one autocorrelation coefficient (0.20). We observe the large difference of the two autocorrelation functions: that of the Markovian process practically vanishes at lag 4, whereas that of SSS has positive values for lags as high as 100.

In the next step, a second process V_i is constructed by superimposing another Markovian process M_i :

$$V_i = U_i + M_i - \mu \quad (13)$$

Here the process M_i is constructed in a different way, rather than using the AR(1) model, yet without losing its Markovian behavior, so that its autocorrelation is

$$\text{Corr}[M_i, M_{i+j}] = \varphi^j \quad (14)$$

for $\varphi > \rho$. More specifically, a continuous time process M (see explanatory sketch in Fig. 4b) with the following

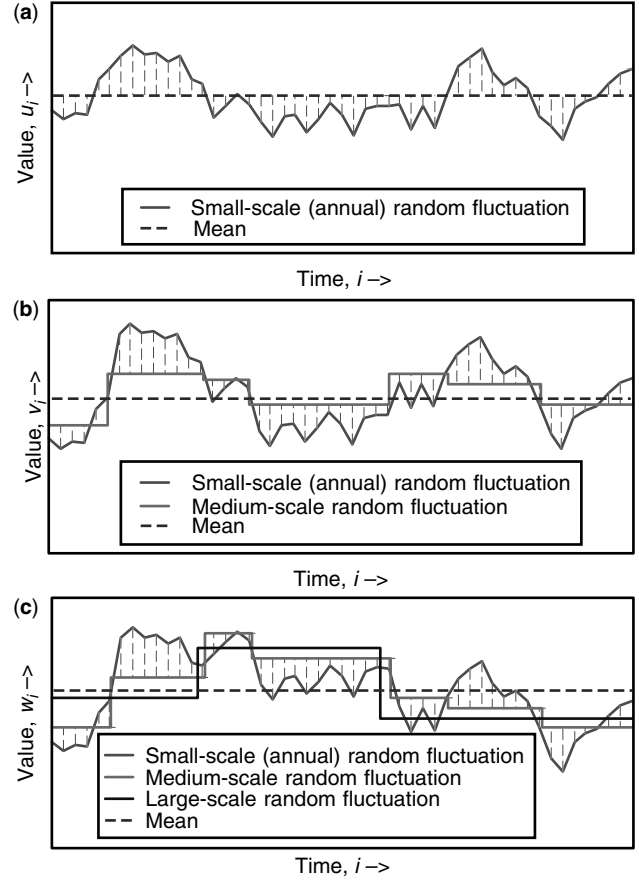


Figure 4. Illustrative sketch of multiple timescale random fluctuations of a process that can explain the Hurst phenomenon: (a) a time series from a Markovian process with constant mean; (b) the same time series superimposed on a randomly fluctuating mean on a medium timescale; (c) the same time series further superimposed on a randomly fluctuating mean on a large timescale (from Reference 8).

properties was assumed: (1) it has mean μ and some variance $\text{Var}[M]$, (2) any realization m of M lasts N years and is independent of previous realizations, and (3) N is a random variable exponentially distributed with mean $\lambda = -1/\ln \varphi$. (This means that N can take nonintegral values). In other words, M takes a value $m_{(1)}$ that lasts n_1 years, then it changes to a value $m_{(2)}$ that lasts n_2 years, etc. (where the values $m_{(1)}, m_{(2)}, \dots$ can be generated from any distribution). The exponential distribution of N indicates that the points of change are random points in time. If we denote M_i the instance of the M process at discrete time i , it can be shown that M_i is Markovian with lag one autocorrelation φ . This way of constructing M_i allows us to interpret V_i as a process similar to U_i but with mean M_i that varies randomly in time (rather than being constant, μ) shifting among randomly determined values $m_{(1)}, m_{(2)}, \dots$, each lasting a random time period with average λ . It can be easily shown from Eq. 13 that the autocorrelation of V_i for lag j is

$$\text{Corr}[V_i, V_{i+j}] = (1 - c)\rho^j + c\varphi^j \quad (15)$$

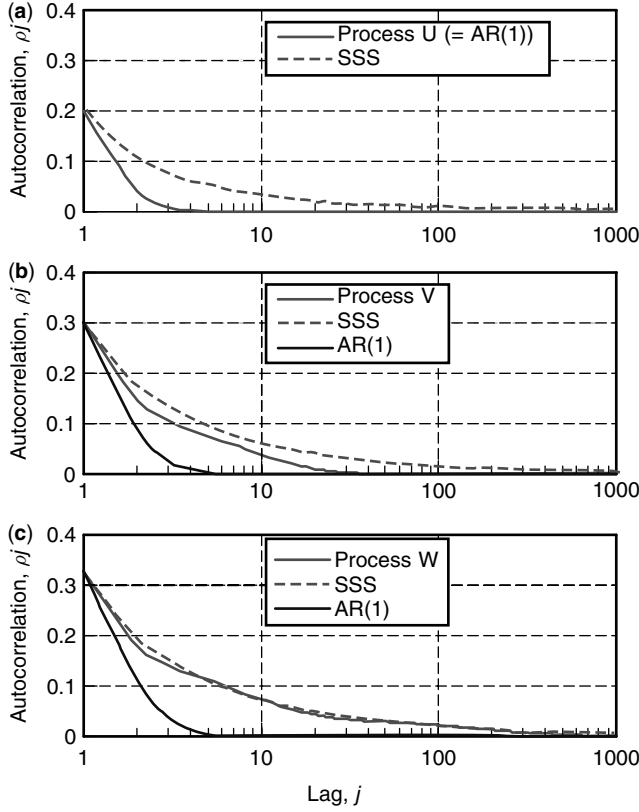


Figure 5. Plots of the example autocorrelation functions of (a) the Markovian process U with constant mean; (b) the process U superimposed on a randomly fluctuating mean on a medium timescale (process V); (c) the process U further superimposed on a randomly fluctuating mean on a large timescale (process W). The superimposition of fluctuating means increases the lag one autocorrelation (from $\rho_1 = 0.20$ for U to $\rho_1 = 0.30$ and 0.33 for V and W , respectively) and also shifts the autocorrelation function from the AR(1) shape (also plotted in all three panels) toward the SSS shape (also shown in all three panels) (from Reference 8).

where $c := \text{Var}[M_i] / (\text{Var}[M_i] + \text{Var}[U_i])$. Setting, for instance, $\lambda = 7.5$ years ($\varphi = 0.875$) and $c = 0.146$, we get the autocorrelation function shown in Fig. 5b, which has departed from the AR(1) autocorrelation and approached the SSS autocorrelation.

In a third step, another process W_i is constructed by superimposing on V_i a third Markovian process P_i :

$$W_i = V_i + P_i - \mu = U_i + M_i + P_i - 2\mu \quad (16)$$

P_i is constructed in a way identical to that of M_i , but with lag one autocorrelation $\xi > \varphi$, so that the mean time between changes of the value of P is $\nu = -1/\ln \xi$. Working as in the previous step, we find

$$\text{Corr}[W_i, W_{i+j}] = (1 - c_1 - c_2)\rho^j + c_1\varphi^j + c_2\xi^j \quad (17)$$

where c_1 and c_2 are positive constants (with $c_1 + c_2 < 1$). Setting, for instance, $\lambda = 7.5$ years ($\varphi = 0.875$), $\nu = 200$ years ($\xi = 0.995$), $c_1 = 0.146$, and $c_2 = 0.036$, we get the autocorrelation function shown in Fig. 5c, which

has now become almost indistinguishable from the SSS autocorrelation for time lags from 1 to 1000.

In conclusion, a Markovian underlying process can result in a nearly SSS process if there occur random fluctuations of the mean of the process on two different scales (e.g., 7.5 and 200 years), yet the resulting composite process is stationary. If we consider that fluctuations occur on a greater number of timescales, the degree of approximation of the composite process to the SSS process will be even better and can cover time lags greater than 1000 (although the extension to lags beyond 1000 may not have any practical interest in hydrology). In conclusion, the *irregular* changes of climate that, according to the National Research Council (25, p. 21), occur on all timescales can be responsible for and explain the Hurst phenomenon.

This demonstration bridges several ideas that had been proposed to explain the Hurst phenomenon, rather than being a novel explanation. As already discussed, it is similar to Klemes's model 4, except for the setting of multiple timescales for the fluctuation of the mean and the emphasis on the stationarity of the composite process. Here, note that Klemes referred to all his "changing mean" models as models with nonstationarity in their mean, even though this is strictly true only for models 1 and 2. He did point out that his final models in group 4 were in fact stationary and that he kept the term "nonstationary" for all changes in the mean to communicate the fact (elaborated in more detail in Ref. 26) that one cannot tell the difference from the pattern of a single "nonstationary-looking" time series (which even a stationary model is designed to mimic), but his explanation has sometimes been missed and led to a misconception about his work by some authors (including this one, who expresses his apology).

The idea of irregular sporadic changes in the mean of the process appeared also in Salas and Boes (27) but not in connection with SSS and not in the setting of multiple timescales. The idea of composite random processes with two timescales of fluctuation appeared in Vanmarcke (28, p. 225). The idea of an explanation of the Hurst phenomenon as a mixture of scales appears in Mesa and Poveda (19). The idea of representing SSS as an aggregation of short-memory processes is the principle of the well known fast fractional Gaussian noise algorithm (FFGN, 29) and is also studied as a possible physical explanation of the Hurst phenomenon by Beran (17, p. 14). The difference in the above described explanation is the aggregation of only three short-memory processes.

IMPORTANCE OF THE HURST PHENOMENON

The presence of the Hurst phenomenon increases dramatically the uncertainty of climatic and hydrologic processes. If such a process were random and our information on this were based on a sample of size n , then the uncertainty in the long term, which can be expressed in terms of the variance of the estimator of the mean, $\bar{X}$, would be

$$\text{var}[\bar{X}] = \frac{\sigma^2}{n} \quad (18)$$

This offers good approximation for a process with short-term persistence, as well, but it is not valid for a process with long-term persistence. Instead, the following relation holds (5;17, p. 54; 30):

$$\text{var}[\bar{X}] = \frac{\sigma^2}{n^{2-2H}} \quad (19)$$

The difference between Eqs. 18 and 19 becomes very significant for large values of H . For example, in a time series of $n = 100$ years of observations and standard deviation σ , according to the classical statistics (Eq. 18), the standard estimation error, the square root of $\text{var}[\bar{X}]$, is $\sigma/10$. However, for $H = 0.8$, the correct standard error, given by Eq. 19, is $\sigma/2.5$, four times larger. To have an estimation error equal to $\sigma/10$, the required length of the time series would be 100,000 years! Obviously, this dramatic difference induces substantial differences in other common statistics as well (5).

A demonstration of the difference in estimates related to climate is given in Fig. 6. Here, a long climatic time series (992 years) was used that represents the Northern Hemisphere temperature anomalies with reference to the 1961–1990 mean (Fig. 8, up). This series was constructed using temperature sensitive palaeoclimatic multiproxy data from 10 sites worldwide that include tree rings, ice cores, corals, and historical documents (31,32). The time series was studied in relation to the Hurst phenomenon by Koutsoyiannis (5), and it was found that the estimate of the Hurst coefficient is 0.88. In the upper panel of Fig. 6, the point estimates and the 99% confidence limits of the quantiles of the temperature anomalies have been plotted for the probability of nonexceedance, u , ranging from 1 to 99%, assuming a normal distribution, as verified from the time series, and using the classical statistical estimators. This is done for two timescales, the basic one ($k = 1$) that represents the annual variation of temperature anomaly and the 30-year timescale, which typically is assumed to be sufficient to smooth out the annual variations and provide values representative of the climate. (For the latter, the averaged rather than aggregated time series, i.e., $z_i^{(30)}/30$, has been used.)

If classical statistics is used (Fig. 6, upper panel), then it is observed that, due to the large length of the series, the confidence band is very narrow and the point estimates for the basic and the aggregated timescale differ significantly. The variability of climate, as expressed by the distribution of the average at the 30-year timescale, is very low, despite much higher variability on the annual scale. This justifies the saying “Climate is what you expect, weather is what you get.” Things change dramatically, if the statistics based on the hypothesis of long-term persistence (5) are used with $H = 0.88$. This is depicted in the lower panel of Fig. 6, where it is observed that the variation in the 30-year average is only slightly lower than that of the annual values and the confidence band has dramatically widened for both timescales. This could be expressed by paraphrasing the above proverb to read “Weather is what you get, climate is what you get—if you keep expecting for many years.” The consequences of these differences in estimating climatic uncertainty due to natural variability are obviously very significant.

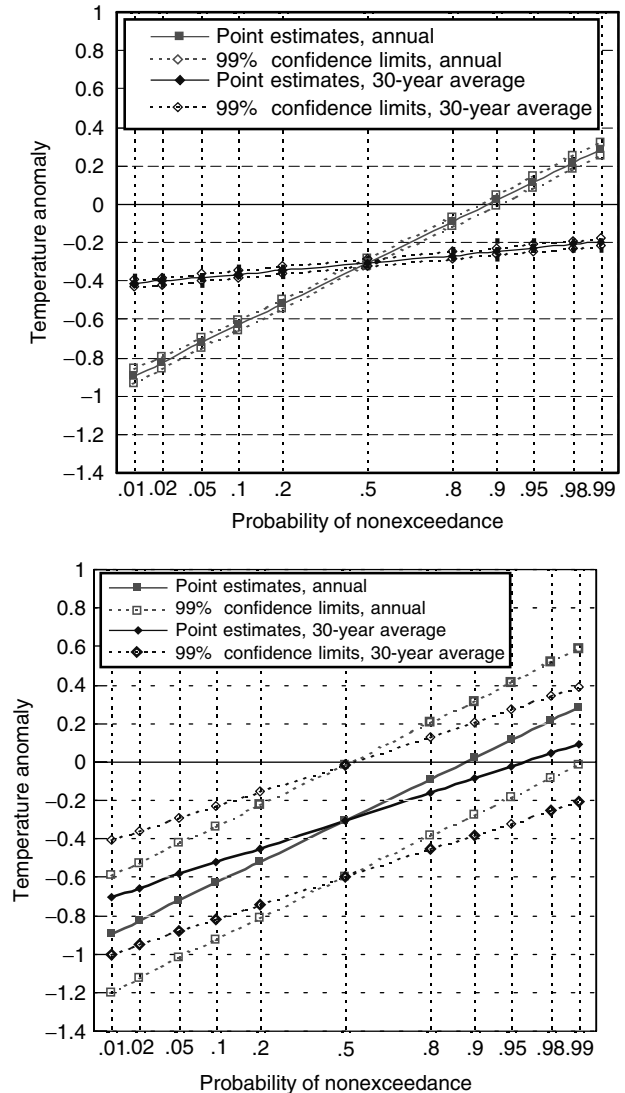


Figure 6. Point estimates of quantiles and 99% confidence limits thereof on the basic timescale (annual values, $k = 1$) and the 30-year timescale (30-year averages, $k = 30$), for the Jones's time series of the Northern Hemisphere temperature anomalies: (up) using classical statistics; (down) using adapted statistics.

The consequences in water resources engineering and management are even more significant. Particularly, because the notion of climate implies long timescales, it is to be expected that the practical importance of the Hurst phenomenon increases in projects whose operating cycles span long periods of time. A typical example may be large reservoirs with multiyear flow regulation (33; see also article SW-776, Reliability Concepts in Reservoir Design). For small-to-middle range reservoirs, it is generally regarded that the effect of the Hurst phenomenon appears to be within the margin of error of hydrologic data used for their design and operation. However, even in hydrosystems with small reservoirs or no reservoirs at all, as becomes obvious from the previous discussion, the effect on the Hurst phenomenon is significant if the uncertainty (not only the expected value) of water availability is to be assessed.

SIMPLE ALGORITHMS TO GENERATE TIME SERIES REPRODUCING THE HURST PHENOMENON

Several algorithms have been developed to generate time series that reproduce the Hurst phenomenon. Among these, we discuss here the simplest ones that can be applied, even in a spreadsheet. These are based on the previously discussed properties of SSS and can be used to provide good approximations of SSS for practical hydrologic purposes.

A first, rather “quick and dirty” algorithm can be very easily formulated based on the deterministic double tent map (Eqs. 10 and 11). The problem with the resulting time series is that consecutive generated values are too regularly and smoothly related. This can be avoided by discarding some of the generated values x_i and holding only the values x_{vj} , for some $v > 1$ and for $j = 1, 2, \dots, n$, where n is the required series length. Figure 7 depicts the attained Hurst coefficient in a time series generated from the double tent map (Eq. 10) either untransformed or transformed (Eq. 11) for $v = 1$ and 4. This figure can serve as a tool to estimate the parameter κ required to achieve a certain Hurst coefficient H (assuming $\lambda = 0.001$). A time series so generated can then be transformed linearly to acquire the required mean and standard deviation.

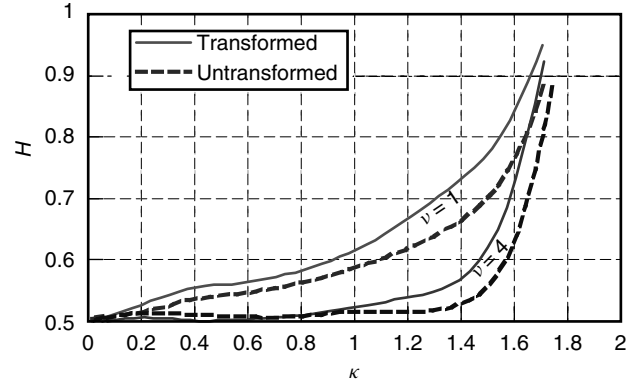


Figure 7. Hurst coefficient of a time series generated from the double tent map (Eqs. 10 and 11) for parameter values $\lambda = 0.001$, κ ranging from 0 to 2, and $v = 1$ and 4. For κ approaching 2, the double tent map has runaway behavior.

By appropriately choosing the initial values α_0 and u_0 , one can obtain a time series that can have a presumed general shape; this requires applying a random search optimization technique. An example of the application of this algorithm to the Jones data set already discussed is depicted in Fig. 8.

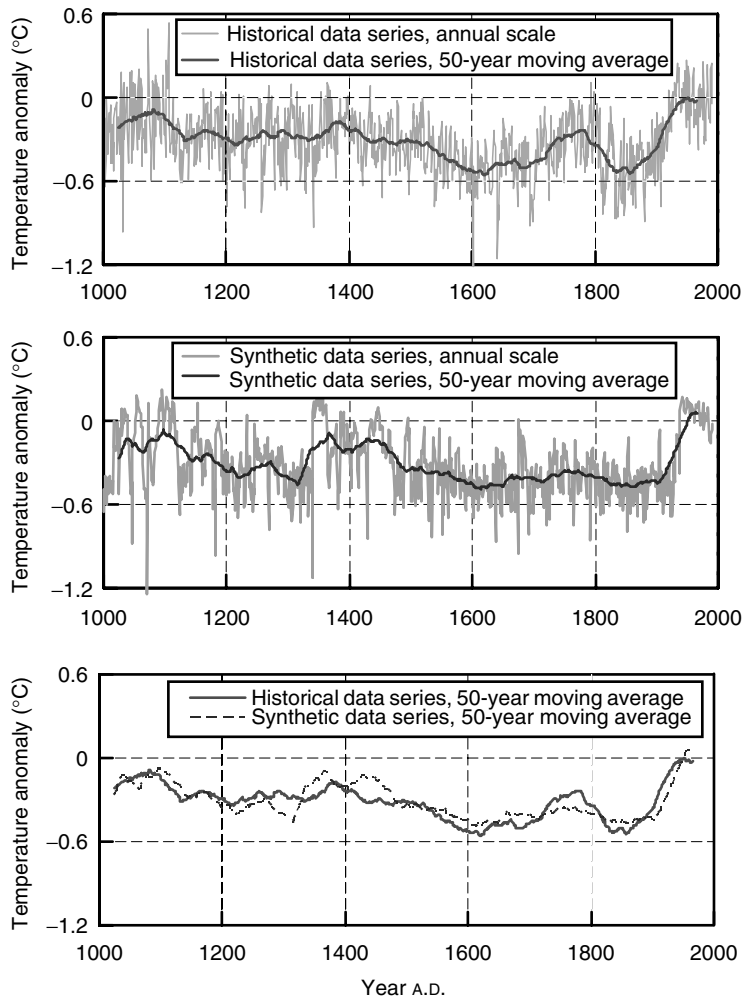


Figure 8. (Up) Plot of the Jones data series indicating Northern Hemisphere temperature anomalies with reference to the 1961–1990 mean; (middle) a synthetic time series generated by the double tent map fitted to the Jones data set and assuming $v = 4$; (down) comparison of the synthetic and original time series in terms of their 50-year moving averages.

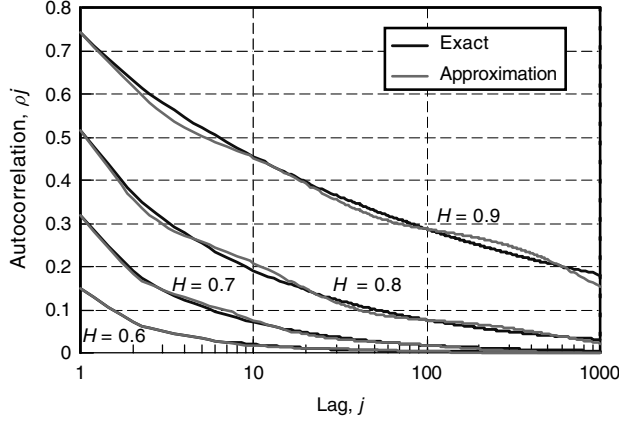


Figure 9. Approximate autocorrelation functions based on Equations 17 and 20–21 vs. the exact SSS autocorrelation functions (Eq. 5) for various values of the Hurst exponent H (from Reference 8).

As we saw earlier, the weighted sum of three exponential functions of the time lag (Eq. 17) can give an acceptable approximation of the SSS autocorrelation function on a basic timescale. This observation leads to an easy algorithm to generate SSS. The following equations (from Reference 8) can be used to estimate the parameters ρ , φ and ξ :

$$\begin{aligned} \rho &= 1.52 (H - 0.5)^{1.32} \\ \varphi &= 0.953 - 7.69 (1 - H)^{3.85} \end{aligned} \quad (20)$$

$$\xi = \begin{cases} 0.932 + 0.087 H & H \leq 0.76 \\ 0.993 + 0.007 H & H > 0.76 \end{cases} \quad (21)$$

The remaining parameters c_1 and c_2 can then be estimated so that the approximate autocorrelation function (Eq. 17) matches the exact function (Eq. 5) for two lags, for example, lags 1 and 100. (Their values are obtained by solving two linear equations). Comparison plots of approximate autocorrelation functions based on Equations 17 and 20–21 versus the exact SSS autocorrelation functions (Eq. 5) for various values of the Hurst exponent H are shown in Fig. 9. Equations 16 and 17 may be interpreted as representing the sum of three independent AR(1) processes, with lag one correlation coefficients ρ , ϕ , and ξ , and variances $(1 - c_1 - c_2) \gamma_0$, $c_1 \gamma_0$, and $c_2 \gamma_0$, respectively. Thus, the generation algorithm is as simple as the generation of three AR(1) series and their addition.

The simple expressions of the statistics of the aggregated SSS process make possible a disaggregation approach for generating SSS (8). Specifically, let us assume that the desired length n of the synthetic series to be generated is 2^m where m is an integer (e.g., $n = 2, 4, 8, \dots$); if not, we can increase n to the next power of 2 and then discard the redundant generated items. We first generate the single value of $Z_1^{(n)}$ knowing its variance $n^{2H} \gamma_0$ (from Eq. 3). Then we disaggregate $Z_1^{(n)}$ into two variables on the timescale $n/2$, $Z_1^{(n/2)}$ and $Z_2^{(n/2)}$, and we proceed this way until the series $Z_1^{(1)} \equiv X_1, \dots, Z_n^{(1)} \equiv X_n$ is generated (see explanatory sketch in Fig. 10).

We consider the generation step in which we disaggregate the higher level amount $Z_{2^{i-1}}^{(k)}$ ($1 < i < n/k$) into two lower level amounts $Z_{2^{i-1}}^{(k/2)}$ and $Z_{2^i}^{(k/2)}$ so that

$$Z_{2^{i-1}}^{(k/2)} + Z_{2^i}^{(k/2)} = Z_{2^{i-1}}^{(k)} \quad (22)$$

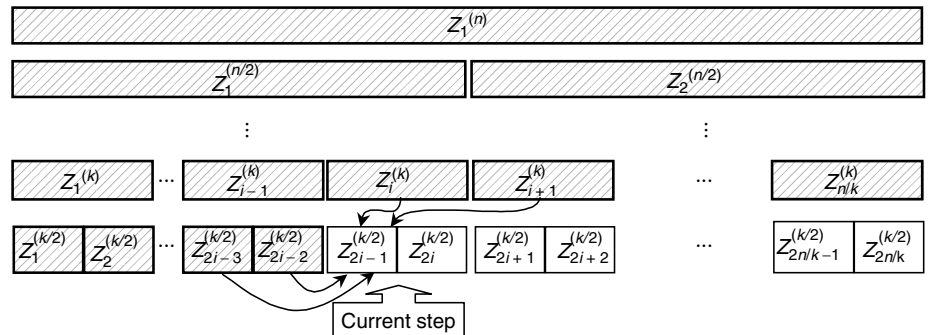
Thus, it suffices to generate $Z_{2^{i-1}}^{(k/2)}$ and then obtain $Z_{2^i}^{(k/2)}$ from Eq. 22. At this generation step, we have available the already generated values of previous lower level time steps, $Z_1^{(k/2)}, \dots, Z_{2^{i-2}}^{(k/2)}$, and of next higher-level time steps, $Z_{2^{i-1}}^{(k)}, \dots, Z_{2^{i-1}}^{(k)}$ (see Fig. 10). Theoretically, it is necessary to preserve the correlations of $Z_{2^{i-1}}^{(k/2)}$ with all previous lower level variables and all next higher level variables. However, we can get a very good approximation if we consider correlations with only one higher level time step behind and one ahead. Under this simplification, $Z_{2^{i-1}}^{(k/2)}$ can be generated from the linear relationship,

$$Z_{2^{i-1}}^{(k/2)} = a_2 Z_{2^{i-3}}^{(k/2)} + a_1 Z_{2^{i-2}}^{(k/2)} + b_0 Z_{2^{i-1}}^{(k)} + b_1 Z_{2^{i+1}}^{(k)} + V \quad (23)$$

where a_2, a_1, b_0 and b_1 are parameters given by

$$\begin{bmatrix} a_2 \\ a_1 \\ b_0 \\ b_1 \end{bmatrix} = \begin{bmatrix} 1 & \rho_1 & \rho_2 + \rho_3 & \rho_4 + \rho_5 \\ \rho_1 & 1 & \rho_1 + \rho_2 & \rho_3 + \rho_4 \\ \rho_2 + \rho_3 & \rho_1 + \rho_2 & 2(1 + \rho_1) & \rho_1 + 2\rho_2 + \rho_3 \\ \rho_4 + \rho_5 & \rho_3 + \rho_4 & \rho_1 + 2\rho_2 + \rho_3 & 2(1 + \rho_1) \end{bmatrix}^{-1} \cdot \begin{bmatrix} \rho_2 \\ \rho_1 \\ 1 + \rho_1 \\ \rho_2 + \rho_3 \end{bmatrix} \quad (24)$$

Figure 10. Explanatory sketch of the disaggregation approach for generating SSS. Gray boxes indicate random variables whose values have been already generated prior to the current step, and arrows indicate the links to those of the generated variables that are considered in the current generation step (from Reference 8).



with ρ_j given by Eq. 5 and V is an innovation with variance,

$$\text{Var}[V] = \gamma_0^{(k/2)} (1 - [\rho_2, \rho_1, 1 + \rho_1, \rho_2 + \rho_3][a_2, a_1, b_0, b_1]^T) \quad (25)$$

where the superscript T denotes the transpose of a vector.

All parameters are independent of i and k , and therefore they can be used in all steps. When $i = 1$, there are no previous time steps, and thus the first two rows and columns of the above matrix and vectors are eliminated. Similarly, when $i = n/k$, there is no next time step, and thus the last row and column of the above matrix and vectors are eliminated.

The power law of the power spectrum of SSS allows generating an SSS time series X_i by filtering a series of white noise V_i using the symmetrical moving average (SMA) scheme (34):

$$X_i = \sum_{j=-q}^q a_{|j|} V_{i+j} = a_q V_{i-q} + \dots + a_1 V_{i-1} + a_0 V_i + a_1 V_{i+1} + \dots + a_q V_{i+q} \quad (26)$$

where q theoretically is infinity but in practice can be restricted to a finite number, as the sequence of weights a_j tends to zero for increasing j . Koutsoyiannis (8) showed that the appropriate sequence of a_j is

$$a_j \approx \frac{\sqrt{(2-2H)\gamma_0}}{3-2H} (|j+1|^{H+0.5} + |j-1|^{H+0.5} - 2|j|^{H+0.5}) \quad (27)$$

The sequence length q must be chosen at least equal to the desired number of autocorrelation coefficients m that are to be preserved. In addition, the ignored terms a_j beyond a_q must not exceed an acceptable tolerance $\beta\sigma$. These two conditions result in

$$q \geq \max \left[m, \left(\frac{2\beta}{H^2 - 0.25} \right)^{1/(H-1.5)} \right] \quad (28)$$

Thus, q can be very large (on the order of thousands to hundreds of thousands) if H is large (e.g., > 0.9) and β is small (e.g., < 0.001). Approximate autocorrelation functions based on Eqs. 26 and 27 versus the exact SSS autocorrelation functions (Eq. 5) for various values of H and q are shown in Fig. 11.

This method can also generate non-Gaussian series with skewness ξ_X by appropriately choosing the skewness of the white noise ξ_V . The relevant equations for the statistics of V_i , which are direct consequences of Eq. 26, are

$$\begin{aligned} \left(a_0 + 2 \sum_{j=1}^s a_j \right) E[V_i] &= \mu, \\ \text{Var}[V_i] &= 1, \\ \left(a_0^3 + 2 \sum_{j=1}^q a_j^3 \right) \xi_V &= \xi_X \gamma_0^{3/2} \end{aligned} \quad (29)$$

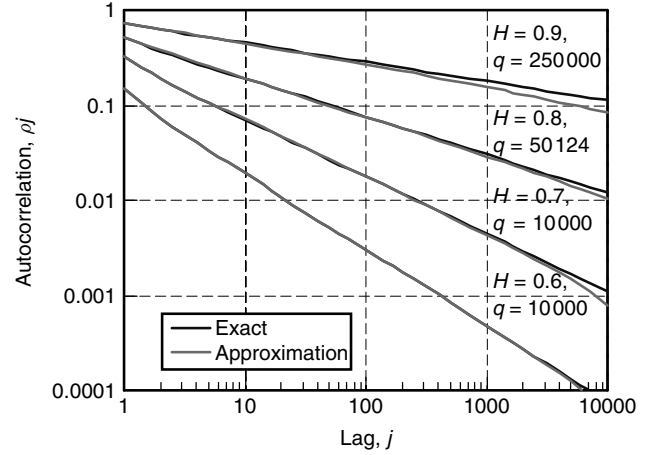


Figure 11. Approximate autocorrelation functions based on Equations 26 and 27 vs. the exact SSS autocorrelation functions (Eq. 5) for various values of the Hurst exponent H and of the number of weights q (from Reference 8).

CONCLUDING REMARKS

More than half a century after its discovery, the Hurst phenomenon has been verified as almost omnipresent in several processes in nature (e.g., hydrology), technology (e.g., computer networks), and society (e.g., economics). However, its consequences are still not widely understood or are ignored; to quote Klemeš (20), it is still regarded by many as “a ghost to be conjured away.”

For example, in stochastic hydrologic simulations that are used in hydrosystem modeling, the Hurst phenomenon is generally not reproduced. The most widespread stochastic hydrology packages have not implemented any types of models that reproduce the Hurst phenomenon. However, today methodologies exist, implemented in software packages, which can reproduce the Hurst phenomenon even in complicated situations, such as a multivariate setting with multiple timescales and asymmetric probability distributions (15,34,35). In addition, as described before, reproducing the Hurst phenomenon in univariate problems is quite simple.

In hydrologic analysis, it has been a common practice to detect falling or rising monotonic “trends” in available records, assume that these are deterministic components, and then “subtract” them from the time series to obtain a “detrended” time series, which is finally used in subsequent analyses. This common technique, which is described in several hydrologic texts, obviously contradicts the Hurst phenomenon. The “trends” are large scale fluctuations, the basis of the Hurst phenomenon. They could be regarded as deterministic components if a sound, physically based model could capture them and also predict their evolution in the future. This, however, is not the case. The a posteriori fitting of a regression curve (e.g., a linear equation) on historical data series has no relation to deterministic modeling. The subtraction of the “trends” from the time series results in a reduction of the standard deviation, an artificial decrease of uncertainty. This is exactly opposite to the real meaning of the

Hurst phenomenon, which, as analyzed before, increases uncertainty substantially.

Even without adopting this “detrending” technique, hydrologic statistics, the branch of hydrology that deals with uncertainty, in its current state is not consistent with the Hurst phenomenon. Typical statistics used in hydrology such as means, variances, cross- and auto-correlations and Hurst coefficients, and the variability thereof, are based on classical statistical theory, which describe only a portion of natural variability, and thus its results may underestimate dramatically the natural uncertainty and the implied risk.

The situation is even worse in climatology, which again uses the classical statistical framework but on longer timescales (e.g., 30 years). As demonstrated before, the consequences of the Hurst phenomenon for natural variability increase as the timescale increases. Recently, many researchers are involved in detecting anthropogenic climatic changes mostly using classical statistical tests, without taking into account the Hurst phenomenon. If statistical estimators consistent to the Hurst phenomenon are used, a choice more consistent with nature, it is more unlikely that such tests will result in statistically significant changes.

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UNIT HYDROGRAPH

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On the scale of a hydrologic event, a river basin schematically has two influences. The first is the yield of rainfall transformed into effective rainfall (in terms of runoff production), and the second is the transfer of runoff flows generated all over the basin territory to the unique outlet. The concept assumes that all transfer processes of the whole basin can be synthesized in a single linear function, called the unit hydrograph of the basin considered (1). It is at the root of the idea of the transfer function and is a kind of functional signature of the basin (2). This means that a characteristic function can be identified for a given basin. It is assumed that this function underlies any rainfall–runoff event and synthesizes all relevant hydrologic processes and geographic influences on the temporal scale of the event and the spatial scale fixed by the basin territory. Furthermore this means that the unit hydrograph should really be the hydrograph at the outlet, following an average impulse unit effective rainfall input, and that the response of the basin to any actual effective rainfall chronicle can be simulated through the sum of successive elementary affinities of the unit hydrograph. Such a holistic purpose necessarily faces problems of simplifying the natural complexity and of identifying the unit hydrograph itself.

The mathematical translation of the unit hydrograph concept is based on the following assumptions and analytical proposals:

- One considers a 1-unit height of effective rainfall generated uniformly over the basin territory during a time-unit duration.
- The response at the outlet of the basin is the characteristic unit hydrograph $u(t)$ during time t .

- This analytical function is used as a linear operator, which means that it respects the affinity and additivity properties. The response to any height of uniform unit effective rainfall is proportional to the unit hydrograph. The response to two following unit effective rainfalls is the sum of two following unit hydrographs. These two properties allow calculating the discharge chronicle $Q(t)$ generated by a chronicle of effective rainfall $\overline{R}_{\text{eff}}(t)$:

$$Q(t) = \frac{S}{\Delta t} \cdot \sum_{\tau=1}^{\min(t, \theta)} \overline{R}_{\text{eff}}(t - \tau + 1)u(\tau) \quad (1)$$

- where
- Δt : the discrete time pace;
 - $Q(t)$: the mean discharge during the time step $[t - \Delta t; t]$;
 - $\overline{R}_{\text{eff}}(t)$: the height of effective rainfall, average in space, during the time step $[t - \Delta t; t]$;
 - $u(t)$: the value of the unit hydrograph during the time step $[t - \Delta t; t]$;
 - S : the basin surface area;
 - θ : the length of the unit hydrograph;
 - τ : the time abscissa of the unit hydrograph.

If one considers Δt close to zero, the effective rainfall input is similar to a Dirac function, and the consequent unit hydrograph is then called an instantaneous unit hydrograph. Equation 1 can then be expressed continuously as a convolution:

$$Q(t) = S \overline{i}_{\text{eff}}(t) * u(t) \quad (2)$$

$$\Leftrightarrow Q(t) = S \int_0^t \overline{i}_{\text{eff}}(t - \tau)u(\tau)d\tau \quad (3)$$

where $\overline{i}_{\text{eff}}(t)$ is the intensity of the effective rainfall, average in space, at time t . $u(t)$ is then the kernel function of the convolution.

The initial unit hydrograph concept strongly simplifies the complexity of basin hydrology:

- Its globality does not easily allow accounting for space heterogeneities and variabilities.
- It applies only to rapid components of runoff.
- Its linearity and stationarity can be criticized because actual hydrologic events are nonlinear.
- The claim of describing all transfer processes within a basin is strong regarding the differences between hillslope and channel processes.

But these problems have been studied through different proposals for identifying it.

The analytical expression for the unit hydrograph can be deduced from a comparison of actual rainfall and discharge gauging (3,4). It can also be deduced from conceptual assumptions, based mostly on the concepts of linear reservoir and linear channel (3). The most famous conceptual proposal is Nash's model (5,6), based on an analogy between a basin and n identical linear reservoirs

in series. The unit hydrograph obtained is the following gamma law:

$$u(t) = \frac{1}{k(n-1)!} \left(\frac{t}{k}\right)^{n-1} e^{-\frac{t}{k}} \quad (4)$$

where the parameters n and k have to be calibrated, but where it is a priori assumed that the gamma law is the shape of the unit hydrograph.

To show the influence of basin geomorphology on unit hydrograph identification, a theory has been proposed by Rodriguez-Iturbe and Valdès (7). They define the unit hydrograph as the probability density function of the water travel time to the outlet. They estimate it by combining the probabilities of paths through the Strahler states and the probabilities of residence times in the different Strahler states of the river network. The result is called the geomorphological unit hydrograph.

Parallel to this, some models are based on the isochrone notion. It is already the case for the rational method used since the nineteenth century (8) and is coming back in models based on the geomorphological area and width functions because they are easily available from geographic databases and GIS processing (9,10). This geomorphological basement of the unit hydrograph identification opens, today, new prospects because it allows splitting transfers through hillslopes and through a river channel (11), considering nonlinear transfer processes through a stable geometric structure (8), and accounting for variability of effective rainfall input along the isochrone areas (12).

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HYDROLOGICAL PROCESSES AND MEASURED POLLUTANT LOADS

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HYDROLOGICAL PROCESSES

Hydrological processes involved in the circulation of water between the atmosphere, the land surface, and underground and its return back to the atmosphere is known as the hydrologic cycle (Fig. 1). In the circulation process, water vapors are transported by winds and air currents through the atmosphere (1). When the air mass cools sufficiently, the water vapor condenses into clouds, and a portion falls to the ground as precipitation in the form of snow, rain, sleet, or hail. Water that falls to the ground as precipitation follows many paths on its way back to the atmosphere. The water may be intercepted and taken up by the plants; it may be stored in small depressions or lakes; it can infiltrate the soil; or it can flow over the surface to a nearby stream channel. The sun may cause the water to evaporate directly back into the atmosphere, or the force of gravity may pull it down through the pores of the soil to be stored for years as slowly moving groundwater. Some of the water flowing through the ground returns to the surface to supply water to springs, lakes, and rivers. Water on the ground surface, in streams, or in lakes can return to the atmosphere as vapor through the process of evaporation. Water used by plants may return to the atmosphere as vapor through transpiration, which occurs when water passes through the leaves of plants. Collectively known as evapotranspiration, both evaporation and transpiration occur in greatest amounts during periods of high temperatures and wind, dry air, and sunshine.

This movement of water supports life on the earth and is mainly governed by the energy of the sun, the force of gravity, land use/land cover pattern, soil type, and geological characteristics. The quantities of water in the atmosphere, soils, groundwater, surface water, and other components are constantly changing because of the dynamic nature of the hydrological processes. The magnitude of various storage components including soil water, snow packs, lakes, reservoirs, and rivers can be altered by human activities. With the water

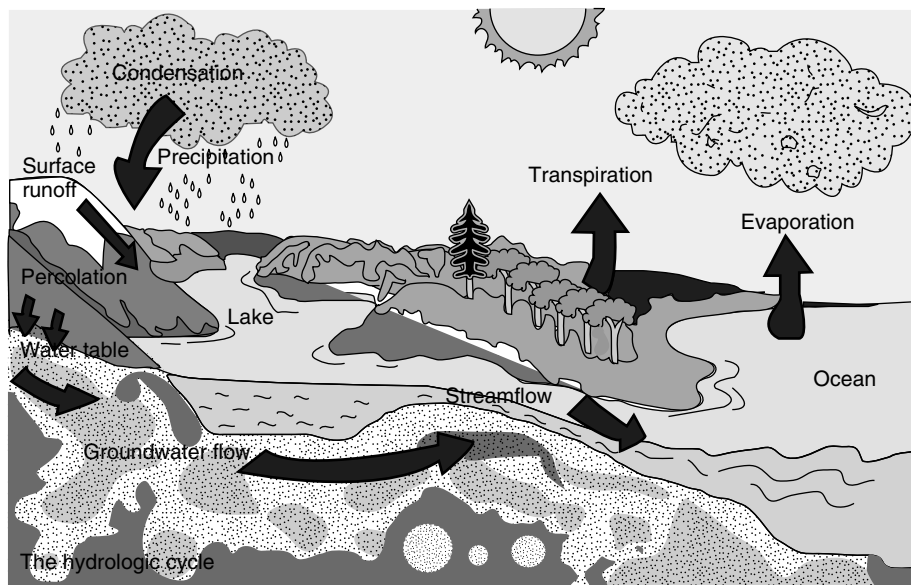


Figure 1. Circulation of water—the hydrologic cycle.

budget approach, experts can examine existing watershed systems, quantify the effects of management impacts on the hydrologic cycle, and, in some cases, predict or estimate the hydrologic consequences of proposed activities. Further, the budget can be estimated in a simplified manner, if components are categorized into input, output, and storage (2). Based on the principle of conservation of mass, inputs such as rainfall, snowmelt, and condensation must balance with changes in storage and outputs, which include stream flow, groundwater seepage, and evapotranspiration (outflow subtracted from the inflow is equal to change in storage). This hydrologic balance, or water budget, is an application of the conservation of mass law expressed by the equation of continuity:

$$I - O = \Delta S \quad (1)$$

where I = inflow, O = outflow, and ΔS = change in storage.

It is to be noted that Eq. 1 does not consider the quality aspects in water budgeting using conservation of mass theory. As a result of increasing population, growth steady rise in irrigation activities, rapid expansion of industries, urbanization, and modern ways of living, tremendous pressure on the available water has been exerted, which is highly uneven in its spatial and temporal distribution both in quantity and quality. Therefore, the water pollution is a major environmental concern all over the world. In the following section, pollutant loads in hydrological variables generated and altered because of human activities have been presented.

MEASURED POLLUTANT LOADS

It is essential for water resources planners and managers to look for water pollution generated in different variables of hydrologic cycle because of human activities. It has been found that among various hydrological variables,

rainfall, runoff (including snowmelt runoff), seepage, and groundwater flow are affected, and the water quality has been degraded beyond permissible limits in many instances.

Pollution Measurement in Rainfall

Acids are the main source of pollution in the rainfall. They occur naturally or because of human activities. Both the phenomena are discussed in the following section.

Pollution Because of Natural Phenomena. Natural sources of acids exist, such as volcanoes, natural geysers, and hot springs. Nature has developed ways of recycling these acids by absorbing and breaking them down. These natural acids contribute to only a small portion of the acidic rainfall in the world today. In small amounts, these acids actually help dissolve nutrients and minerals from the soil so that trees and other plants can use them for food. The large amounts of acids produced by human activities overload this natural acidity.

Pollution Because of Human Activities. On a daily basis, human activities (industrial, agricultural, and residential) cause vast quantities of natural and synthetic chemicals to be emitted into the atmosphere. Once released, the substances are dispersed throughout the globe by air currents.

Scientists have discovered that air pollution from the burning of fossil fuels is the major cause of acid rain (Fig. 2). Power plants use coal and oil to produce the electricity we need to heat and light our homes and to run our electric appliances. Natural gas, coal, and oil are burnt to heat homes. Cars, trucks, and airplanes use gasoline, another fossil fuel. The smoke and fumes from burning fossil fuels rise into the atmosphere and combine with the moisture in the air to form acid rain (3).

When water droplets form and fall to the earth, they pick up particles and chemicals that float in the air. Even

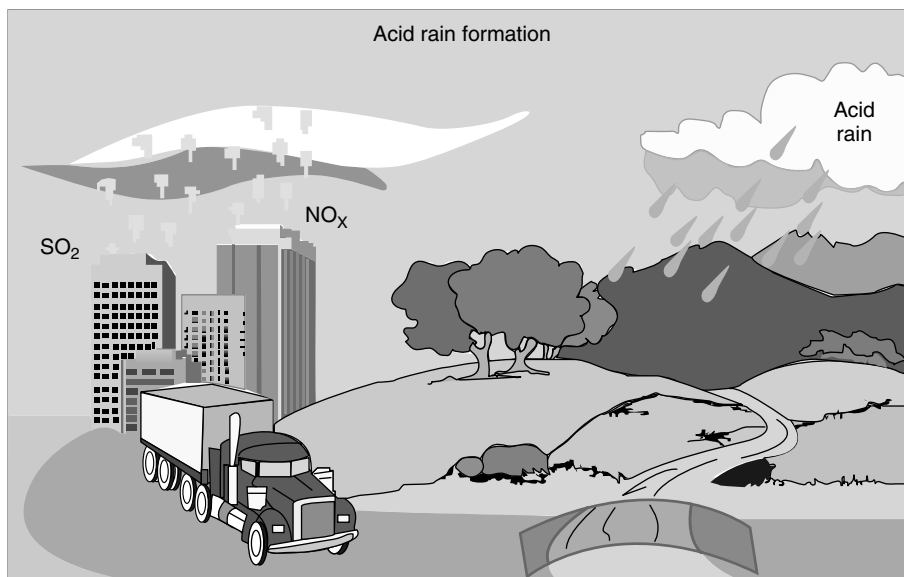


Figure 2. Acid rain formation.

clean, unpolluted air has some particles such as dust or pollen. Clean air also contains naturally occurring gases such as carbon dioxide. The interaction between the water droplets and the carbon dioxide in the atmosphere gives rain a pH of 5.6, making even clean rain slightly acidic. Other natural sources of acids and bases in the atmosphere may lower or raise the pH of unpolluted rain. However, when rain contains pollutants, especially sulfur dioxide and nitrogen oxides, the rain water can become very acidic.

The main chemicals in air pollution that create acid rain are sulfur dioxide and nitrogen oxides. Acid rain usually forms high in the clouds where sulfur dioxide and nitrogen oxides react with water, oxygen, and oxidants, which forms a mild solution of sulfuric acid and nitric acid. Sunlight increases the rate of most of these reactions. Rainwater, snow, fog, and other forms of precipitation containing those mild solutions of sulfuric and nitric acids fall to the earth as acid rain with a consequent decrease of water pH (pH

below 5.5–5.6). Figure 3 depicts the acid rain formation in the atmosphere.

Effects of Acid Rain. Acid rain does not account for all of the acidity that falls back to earth from pollutants. About half the acidity in the atmosphere falls back to the earth through dry deposition as gases and dry particles. The wind blows these acidic particles and gases onto buildings, cars, homes, and trees. In some instances, these gases and particles can eat away the things on which they settle. Dry deposited gases and particles are sometimes washed from trees and other surfaces by rainstorms. When that happens, the runoff water adds those acids to the acid rain, making the combination more acidic than the falling rain alone. The combination of acid rain plus dry deposited acid is called acid deposition.

Acid rain can contaminate drinking water, damage vegetation and aquatic life, and erode buildings and

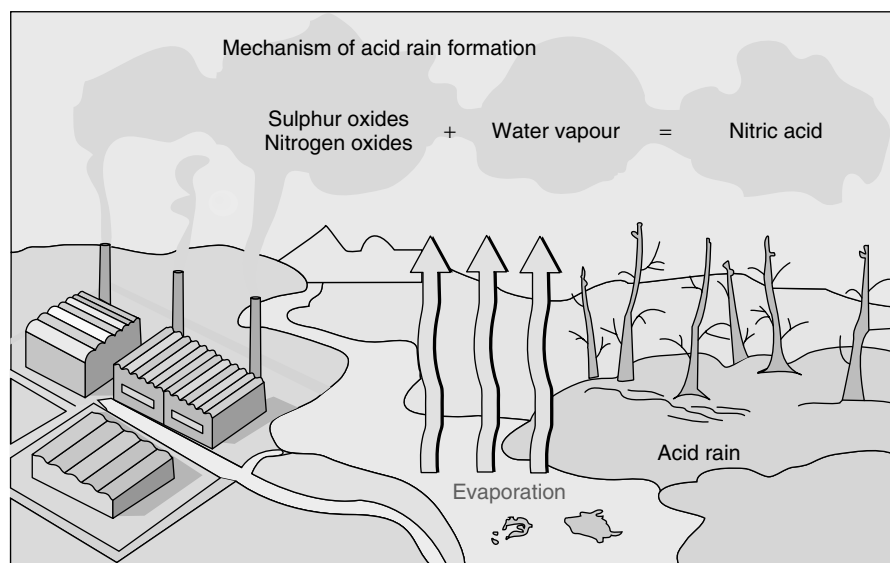


Figure 3. Acid rain formation in the atmosphere.

monuments. The water bodies most susceptible to change because of acid precipitation are those whose catchments have shallow soil cover and poorly weathering bedrock, e.g., granite and quartzite. These soil types are characterized by the absence of carbonates that could neutralize acidity. The runoff water from such areas is less buffered than from areas such as limestone catchments, with an adequate level of carbonate. Such catchments and waters are termed acid-sensitive (poorly buffered), and can suffer serious ecological damage because of artificially acidified precipitation from air masses downwind of major emissions.

Numerous studies show that acid precipitation damages forests and may cause significant decreases in productivity. Acid precipitation causes foliar damage to birch and pines, impairs seed germination of spruce seeds, erodes protective waxes from oak leaves, and leaches nutrients from plant leaves. Scientists believe that damage to trees may increase the likelihood of avalanches and landslides because trees help retain snow cover and soil cover on steep mountainsides.

Concern for agriculture has also been raised by numerous researchers. Acid precipitation is particularly harmful to buds; therefore, acids falling on plants in the spring might impair growth. Acids may also damage plants by altering the soil. For example, acid rain may leach important elements from the soil, resulting in lower yield and reduced agricultural output. Acidification of soils may also impair soil bacteria and fungi that play an important role in nutrient cycling and nitrogen fixation, both essential to normal plant growth.

The toxic action of acidity is most pronounced in its effect on the fish and macroinvertebrate fauna of streams and lakes. Dissolved aluminium concentrations as low as 0.05–1 mg can kill such organisms, whereas lower levels may have nonlethal effects such as respiratory difficulty, impaired growth, and reproductive ability, which, in the long term, may lead to their elimination.

Pollution Measurement in Runoff

The portion of water that does not infiltrate the soil but flows over the surface of the ground to a stream channel is called surface runoff. Water always takes the path of least resistance, flowing downhill from higher to lower elevations, eventually reaching a river or its tributaries. Runoff is the greatest source of water pollution, contributing from 50–60% of the pollutant load. As this water flows over agricultural lands, urban areas, hills, and forest land, it picks up different types of pollutants such as silt particles, microbial contaminants, pesticides, fertilizer, pet waste, litter, industrial wastes, urban wastes, and oil.

Natural Sources of Runoff Pollution. Heavy rain falling on exposed soil can cause substantial leaching of nitrate, some of which goes directly into rivers, but most of which percolates into the groundwater from where it may eventually reach the rivers if no natural denitrification occurs. Assessment of trends in nitrate concentrations should be undertaken on a long-term basis (i.e., frequent sampling for more than 10 years' duration).

In regions of high erosion (e.g., steep slopes, heavy rainfall, highly erodible rocks) where natural total suspended solids (TSS) already exceed 1 gm per litre, intensive agriculture aggravates the natural erosion rates.

Increased mineral salts in rivers may develop from potash mines, salt mines, iron mines, and coal mines. Mining wastes result in increases in specific ions only, such as Cl^- and Na^+ from potash and salt mines, and SO_4^{2-} from iron and coal mine wastes. The changes in ionic contents and the ionic ratio of waters are very often linked to pH changes. Mine wastewaters are generally very acidic ($\text{pH} \leq 3$), whereas industrial wastes may be basic or acidic.

Increased evaporation and evapotranspiration in the river basin (mainly in arid and subarid regions) also increase minerals. Evaporation affects all ions and as calcium carbonate reaches saturation levels, calcium sulphate-rich waters or sodium chloride-rich waters are produced. Increased evaporation and evapotranspiration in the river basin (mainly in arid and subarid regions) also increase minerals. Salinization resulting from evaporation usually leads to more basic pH levels.

In colder regions where snowmelt has a significant hydrological influence, the accumulated acidic deposition in the snow may be released when it melts. Melting snow during spring releases its acid in a sudden torrent, quickly elevating the acidity of lakes and streams. This surge of acids coincides with the sensitive reproductive period for many species of fish. The first 30% of the melt water contains virtually all of the acid and typically has a pH of 3 to 3.5, which is toxic to eggs, fry, and adult fish as well, which can cause a sudden acid pulse, which may be more than one pH unit lower than normal (4).

Runoff Pollution Because of Human Interference. There are many causes for water pollution, but two general categories exist: direct and indirect contaminant sources (Fig. 4). Direct sources include effluent outfalls from factories, refineries, waste treatment plants, etc. that emit fluids of varying quality directly into water supplies.

Indirect sources include contaminants that enter from soils/groundwater systems and from the atmosphere via rain water. Soils and groundwater contain the residue of human agricultural practices (fertilizers, pesticides, etc.) and improperly disposed of industrial wastes (5). Atmospheric contaminants are also derived from human practices (such as gaseous emissions from automobiles,

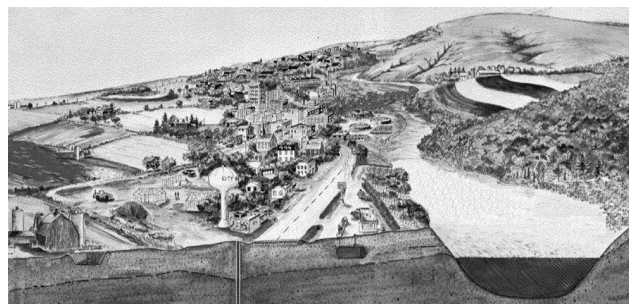


Figure 4. Representation of direct and indirect sources of pollution.

Table 1. Nonpoint Sources Pollution and Pollutants

Category of Nonpoint Source	Pollution Causing Activities	Principal Pollutants
Agriculture	Crop production, animal production, farm production	Nutrients, salts, fertilizer, pesticides, organic matters
Forestry	Access road construction and operation, harvesting systems, logging, crop regeneration, other silvicultural processes	Nutrients, pesticides, organic pollutants
Mining	Exploration, construction of facilities, mine operation, mine abandonment	Nutrients, waters, dissolved minerals, salinity
Construction	Land development, transportation and communication networks, water resources facilities	Chemicals, biological materials
Urban runoff	Precipitation discharges containing pollutants, accelerated and concentrated by urban surfaces and collection system	Organic materials, coliform bacteria, pesticides, nutrients, heavy metals
Hydrologic modifications	Channel modifications, farm drainage, dams, resource recovery, and related activities	Nutrients, pesticides, thermal chemicals, micro-organisms
Residual wastes	Foregoing categories create residual waste not discharged to water but conveyed by runoff and infiltration	Both hazardous and nonhazardous

factories, and even bakeries as discussed earlier). Table 1 reflects the pollution causing activities and their principal pollutants of nonpoint sources of water pollution.

Many physical, chemical, and biological parameters are measured to determine the overall quality of water. However, the quality of water is based not only on the concentration of substances but also on the intended use of the water. A person using water for drinking would have a different set of criteria for determining the water quality compared with a person using the water for swimming. In Tables 2, 3, and 4, the water quality criteria based on the use of water (such as drinking, bathing, and irrigation) is presented (8).

Pollutant Loads in Groundwater

Where water infiltrates the ground, gravity pulls the water down through the pores until it reaches a depth in the ground where all of the spaces are filled with water (Fig. 5). At this point, the soil or rock becomes saturated, and the water level that results is called the water table. The water table is not always at the same depth below the land surface. During periods of high precipitation, the water table can rise. Conversely, during periods of low precipitation and high evapotranspiration, the water table falls. The area below the water table is called the saturated zone, and the water in the saturated zone is called groundwater. The area above the water table is the unsaturated zone.

Groundwater is found in aquifers, which consist of soil or rock in the saturated zone that can yield significant amounts of water. In an unconfined aquifer, the top of the aquifer is defined by the water table. Confined aquifers are bound on the top by impermeable material, such as clay. Water in a confined aquifer is normally under pressure and can cause the water level in a well to rise above the water table. If the water rises above the ground surface, it is designated a flowing artesian well. A perched water table occurs when water is held up by a low permeability material and is separated from a second water table below by an unsaturated zone. In the saturated zone, groundwater flows through the pores of the soil or rock both laterally and vertically.

Water moving from an aquifer and entering a stream or lake is called groundwater discharge, whereas any water entering an aquifer is called recharge. An aquifer may receive recharge from these sources, an overlying aquifer, or more commonly from precipitation followed by infiltration. The pollutants present in soil also move along the water entering the aquifer and pollute the groundwater.

How We Contaminate Groundwater. Any addition of undesirable substances to groundwater caused by human activities is considered to be *contamination* (9). It has often been assumed that contaminants left on or under the ground will stay there. This assumption has been shown to be wishful thinking. Groundwater often spreads the effects of dumps and spills far beyond the site of the original contamination. Groundwater contamination is extremely difficult, and sometimes impossible, to clean up.

Groundwater contaminants come from two categories of sources: point source and distributed, or nonpoint source. They are as follows:

Point sources

The main sources are:

- On-site septic systems
- Leaky tanks or pipelines containing petroleum products
- Leaks or spills of industrial chemicals at manufacturing facilities
- Underground injection wells (industrial waste)
- Municipal landfills
- Livestock wastes
- Leaky sewer lines
- Chemicals used at wood preservation facilities
- Mill tailings in mining areas
- Fly ash from coal-fired power plants
- Sludge disposal areas at petroleum refineries
- Land spreading of sewage or sewage sludge
- Graveyards
- Road salt storage areas

Table 2. General Quality Criteria for Raw Waters Used for Organized Community Water Supplies (Surface and Groundwater)

(a) Primary parameters (frequency of monitoring may be daily and/or even continuous using automatic instruments for few parameters like pH, DO, and Conductivity)

Parameter	Range/limiting value		Note
	Use with Only Disinfection	Use After Conventional Treatment	
pH	6.5–8.5	6–9	To ensure prevention of corrosion in treatment plant and distribution system and interference in coagulation and chlorination
Color, Pt scale, Hz units	<10	10–50	Color may not get totally removed during treatment (not applicable in monsoon period)
Total suspended solids, mg/l	<10	10–50	High SS may increase the cost of treatment
Odor, dilution factor	<3	3–10	May not be easily tackled during treatment to render water acceptable
Dissolved oxygen, (% saturation)	90–110	80–120	
Biochemical oxygen demand, mg/l	<3	3–5	Could cause problems in treatment, larger chlorine demands, and residual taste and odor problems.
Total Kjeldahl' Nitrogen, mg/l	<1	1–3	Same as above
Ammonia, mg/l	<0.05	0.05–1.5	Same as above
Fecal coliform, MPN/100 ml	<200	200–2000	The criteria would be satisfied if, during a period of one week, not more than 5% samples show greater than 20,000 MPN/100 ml, and not more than 20% of samples show greater than prescribed limit
Conductivity, umhos/cm	<1000	1000–2000	High conductivity may cause unpalatable mineral taste and physiological disorders
Chloride, mg/l	<200	200–300	May cause physiological impact and unpalatable mineral taste
Sulphates, mg/l	<100	100–250	May cause digestive abnormality on prolonged consumption
Phosphates, mg/l	<0.1	0.1–0.3	May interfere with coagulation
Nitrate, mg/l	<30	30–50	High nitrate/nitrite may cause methemoglobinemia
Fluoride, mg/l	<1.0	1.0–1.5	Prolonged consumption of water containing high F may cause fluorosis
Surfactants, mg/l	<0.1	0.1–0.2	May impair treatability and cause foaming

Note: There should not be any significant wastewater discharge in the upstream (up to 5 km) of the water intake point.

(b) Additional Parameters for Periodic (Say Monthly/seasonal) Monitoring

Parameters	Desirable	Acceptable	Note
Dissolved iron, mg/l	<0.3	3–5	Higher iron affects the taste of beverages and causes stains
Copper, mg/l	<0.5	0.5–1.0	May result in damage of liver
Zinc, mg/l	<0.1	<5.0	May cause bitter stringent taste
Arsenic, mg/l	<0.002	0.002–0.05	Can cause hyperkeratosis and skin cancer in human beings
Cadmium, mg/l	<0.001	0.001–0.005	Toxic to man
Total-Cr, mg/l	<0.05	<0.05	Toxic at high doses
Lead, mg/l	<0.05	0.04–0.05	Irreversible damage to the brain in children, anaemia, neurological disfunction, and renal impairment
Selenium, mg/l	<0.01	<0.01	Toxic symptoms similar to arsenic
Mercury, mg/l	<0.0002	0.0002–0.0005	Deadly, poisonous, and carcinogenic
Phenols, mg/l	<0.001	<0.001	Toxic and carcinogenic; may also cause major problem of taste and odor
Cyanides mg/l	<0.05	<0.05	Larger consumption may lead to physiological abnormality
Any Polycyclic Aromatic hydrocarbons, mg/l	<0.0002	<0.0002	Carcinogenic
Total Pesticides, mg/l	<0.001	<0.0025	Tend to bioaccumulate and biomagnify in the environment; toxic and carcinogenic

Note: The parameters mentioned need to be examined only when (i) there are known natural sources in the upstream basin region likely to contribute, (ii) there are known problems in respect to the parameter, (iii) there are other well-founded apprehensions.

- Wells for disposal of liquid wastes
- Runoff of salt and other chemicals from roads and highways
- Spills related to highway or railway accidents

- Coal tar at old coal gasification sites
- Asphalt production and equipment cleaning sites

Among the more significant point sources are municipal landfills and industrial waste disposal sites. When either

Table 3. Quality Criteria for Waters of Mass Bathing Reaches

Parameter	Desirable	Acceptable	Note
Fecal coliform, MPN/100 ml	<100	100–1000	If MPN of fecal coliform is noticed to be more than 100/100 ml, then regular tests should be carried out. The criteria would be satisfied if, during a period, not more than 5% samples show greater than 5000 MPN/100 ml, and not more than 20% of samples show greater than 1000/100 ml
pH	6–9	6–9	
Color Pt scale Hz units	No abnormal color	No abnormal color	No color of anthropogenic origin
Mineral oil, mg/l	No film visible	No film visible	
Methylene blue active substances, mg/l	<0.3	0.3–1.0	Skin problem likely
Phenols, mg/l	<0.005	0.005–0.01	Skin problems and odor problem
Transparency (Secchi depth)	>2 m	0.5–2 m	
Biochemical oxygen demand, mg/l	<5	5–8	High organic matter may be associated with coliform pathogens
Dissolved oxygen, % saturation	80–120	60–140	–
Floating matter of any type	Absent	Absent	

Note: No direct or indirect discharge of untreated domestic/industrial wastewater within 5 km upstream.

Table 4. General Water Quality Criteria for Irrigation Waters (6,7)

Parameter	Generally Acceptable	Relaxation for Special Planned (Exceptional Notified) Cases	Note
Conductivity, umhos/cm	<2250	2250–4000	The irrigation water having conductivity more than 2250 $\mu\text{S}/\text{cm}$ at 25 °C may reduce vegetative growth and yield of the crops. It may also increase soil salinity, which may affect its fertility
Fecal coliform, MPN/100 ml	<5000	No limit	No limit for irrigating crops not eaten raw
pH	6–9	5–9.5	Soil characteristics important
Biochemical oxygen demand, mg/l	<100	No limit	Land can absorb organic matter faster than water; however, the conditions of water logging should be avoided. Stagnant water should not persist for more than 24 hours
Floating materials such as wood, plastic, rubber, etc.	Absent	No limit	May inhibit water percolation
Boron, mg/l	<2	<2	Boron is an essential nutrient for plant growth; however, it becomes toxic beyond 2 mg/l
SAR	<26	No limit	SAR beyond 26 may cause salinity and sodicity in the soil. When it exceeds the limit, method of irrigation and salt tolerance of crops should be kept in mind
Total heavy metals mg/l	< 5.0	5.0–15.0	

of these occur in or near sand and gravel aquifers, the potential for widespread contamination is the greatest. Other point sources are individually less significant, but they occur in large numbers all across the country. Some of these dangerous and widespread sources of contamination are septic tanks and leaks and spills of petroleum products and of dense industrial organic liquids.

Septic systems are designed so that some of the sewage is degraded in the tank and some is degraded and absorbed by the surrounding sand and subsoil. Contaminants that may enter groundwater from septic systems include bacteria, viruses, detergents, and household cleaners, which can create serious contamination problems. Despite the fact that septic tanks and cesspools are known sources of contaminants, they are poorly monitored and studied very little.

Nonpoint (Distributed) Sources

Nonpoint sources comprise the other type-generalized discharges of waste whose location cannot be identified. The main sources are agriculture, forestry, mining, construction, urban runoff, hydrologic modifications, and residual wastes. Table 1 reflects the pollution-causing activities and their principal pollutants of nonpoint sources of water pollution.

Contamination can render groundwater unsuitable for use. Scientists also predict that, in the next few decades, more contaminated aquifers will be discovered, new contaminants will be identified, and more contaminated groundwater will be discharged into wetlands, streams, and lakes.

Once an aquifer is contaminated, it may be unusable for decades. The residence time, as noted earlier, can be

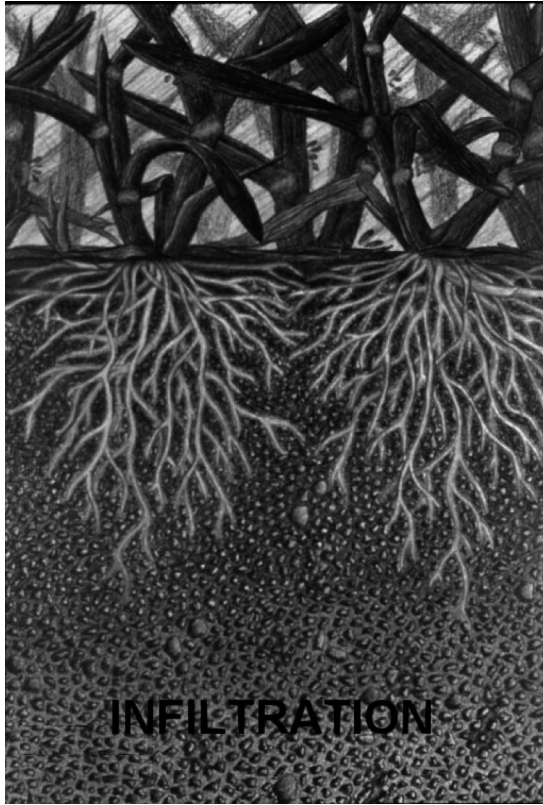


Figure 5. Infiltration of surface water.

anywhere from two weeks to 10,000 years. Furthermore, the effects of groundwater contamination do not end with the loss of well water supplies. Several studies have documented the migration of contaminants from disposal or spill sites to nearby lakes and rivers as this groundwater passes through the hydrologic cycle, but the processes are not as yet well understood. In Canada, pollution of surface water by groundwater is probably at least as serious as the contamination of groundwater supplies. Preventing contamination in the first place is, by far, the most practical solution to the problem. Prevention can be accomplished by the adoption of effective groundwater management practices by governments, industries, and all individuals. Although progress is being made in this direction, efforts are hampered by a serious shortage of groundwater experts and a general lack of knowledge about how groundwater behaves.

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HYDROLOGIC THRESHOLDS

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INTRODUCTION

Hydrologic science has witnessed an exponential growth in the last century, thanks to sheer necessity (or even mere curiosity) and technological advances. Population growth and lifestyle changes have contributed (both positively and negatively) to hydrologic science, in terms of water resources availability, planning, and management. Invention of powerful computers, remote sensing, geographic information systems, worldwide networking facilities, and many other technologies have, in one way or another, allowed better hydrologic data collection, analysis, and sharing, which, in turn, facilitated more effective and efficient water resources planning and management. Despite these developments, our lack of ability to adequately plan and manage our water resources continues to be a matter of grave concern.

Setting aside, for the present discussion, the population growth and the socio-politico-economic scenarios, our inability to adequately plan and manage our water resources may largely be attributed to two factors:

1. our inability to collect all the relevant data for the system of interest; and
2. our inability to understand the past, model the present, and forecast the future, even with collected data; in other words, our inability to accurately “replicate and extrapolate” the data.

In regard to 1, we may take some solace by our experience, albeit limited, that often only a few processes dominate responses in a given hydrologic system and, therefore, may focus on collecting more reliable data relevant only to these dominant processes (1). This

concept, fittingly termed by Grayson and Blöschl (1) as the “Dominant Processes Concept” (DPC), has been receiving increasing attention among hydrologists in one way or another (2–4). More recently, Sivakumar (5) presented a new proposal for moving forward with the DPC, emphasizing on integration of theoretical concepts and practical knowledge for more effective and efficient data collection and analysis procedures.

As for 2, notwithstanding data quantity and quality issues, our inability to accurately replicate and extrapolate the data may largely be because of the inherent limitations of our modeling and forecasting procedures and the inappropriate selection of data sets used therein for “training” (learning or calibration) and “testing” (verification or validation), respectively. Sivakumar (6) points out this “data handling” problem in a rather different and multifaceted manner, by reviewing sample studies on recent data modeling and forecasting techniques in hydrology, including artificial neural networks and chaos theory. More specifically, Sivakumar (6) argues that any bias resulting in our extrapolations is in line with the procedural or data bias in our learning, which is chosen to capture (the statistics of) our desired “extreme” or “average” extrapolations. To overcome this problem, he suggests the use of different models for different ranges of data, rather than a single model for the entire data range, as is commonly used. Deriving an analogy between humans and catchments, and emphasizing their internal characteristics and responses to external events, Sivakumar (6) discusses the usefulness of “thresholds” in identifying these data ranges. Reiterating that the “threshold” concept is already in wide use in different implicit forms in hydrology, he stresses the need for bringing these implicit forms under one common umbrella, and explicitly termed as “thresholds,” for everyone to understand.

In the spirit of the proposal by Sivakumar (6), this article presents further ideas as to the potential role of thresholds in hydrology. Specifically, it aims to launch the threshold concept at a much wider spectrum than the narrow time-series/parameter-optimization spectrum presented by Sivakumar (6), which is attempted by discussing three important issues: (a) relevance of thresholds in hydrology; (b) challenges in the identification and quantification of thresholds; and (c) importance of interdisciplinary efforts on threshold research.

THRESHOLDS IN HYDROLOGY

Simply put, thresholds are “critical” points separating different states (or phases) that exist or are possible in a given system. These points are critical because they are the “enablers” or “barriers” for transitions between states that may have enormous differences in their properties (e.g., physical, chemical, and biological). As these transitions are often “catastrophic”, thresholds are generally defined as points separating “average” and “extreme” events.

The relevance of thresholds in hydrology should be evident even from our simple observations of “average”, “high”, and “low” rainfall periods, which often lead to “normal”, “flood”, and “drought” periods, respectively. Whether it is rainfall, runoff, or any other hydrologic phenomenon,

transitions between different states are essentially the result of the underlying physical mechanisms. These physical mechanisms are, in turn, governed by the characteristics of the land, ocean, and atmosphere. As natural and man-made changes to land, ocean, and atmosphere occur, even within their individual realms, so do changes to the underlying physical mechanisms and transitions from one state to another. When such changes occur across these domains, the “probability” of changes to physical mechanisms and transitions between states increases, because a much more complex coupled land-ocean-atmospheric system is in effect.

The land-ocean-atmospheric system as a whole is a “network” composed of interconnected parts, which in themselves are “networks” (e.g., hydrologic system) composed of their own interconnected parts, and so on. In this network(s), the relationships between the parts are circular (in an abstract, but not necessarily spatial, sense), which causes “feedback” because the actions of each component “feeds back” on itself (e.g., rainfall-evaporation-rainfall or rainfall-runoff-evaporation-rainfall). This feedback mechanism can be either positive or negative, but both types are relevant and important; a negative feedback stabilizes the system, whereas a positive feedback often destabilizes the system.

Feedbacks are not only inherent in hydrologic systems but are also inherently nonlinear in their nature. The nonlinear nature of the feedbacks, when combined with the complex nature of the hydrologic systems, may cause bifurcations, i.e., rapid, unpredictable, and often catastrophic transitions to new states, occurring at critical points or thresholds. One state that can develop at a bifurcation is chaos, which demonstrates sensitive dependence to initial conditions or “butterfly effect” [e.g., (7,8)]. This effect, i.e., infinitesimal changes in inputs leading to enormous differences in outcomes, amplifies uncertainty and limits predictability. In fact, the most complex behavior in hydrologic systems (and other networks) often occur at the edge of chaos, i.e., at a transition from chaos to order or vice-versa. A particular signature of this state is the presence of fractals, i.e., patterns in space and/or time that are self-similar on many scales [e.g., (9)].

A related property of a feedback, nonlinear, chaotic, and fractal system is self-organization, which refers to the spontaneous development (organization) of the system and its evolution to different and more complex states. Self-organizing systems demonstrate the “emergence” of properties. The emergent properties are not inherent in any of the parts of the system examined in isolation, but emerge only as a result of the interactions between the parts that characterize the system as a whole. Self-organization is particularly relevant for hydrologic systems. For example, a river exists as an interaction between the flow of water, the materials making up the riverbank and channel, the elevation and terrain of the land, the climate, temperature and rainfall, the law of gravity, the forces of turbulence, and more. Although it is possible to understand each component of the river by reduction, this is not adequate for understanding the river as a whole, because the emergent properties and evolution

of the river are governed by the interactions among the components, which stretch beyond the domain of any one single component.

IDENTIFICATION AND QUANTIFICATION OF THRESHOLDS

With the relevance and importance of thresholds in understanding hydrologic systems evident, the immediate question is how to identify and quantify them. In this regard, knowledge of general system characteristics (e.g., climatic condition, geography, land use, wet or dry state) and system evolution (i.e., history of changes to the system and the associated changes to hydrologic events, such as rainfall amounts, frequency of floods and droughts) may provide some basic information on the types of thresholds (i.e., identification) and their levels (i.e., quantification). With this information, additional threshold criteria may be established using methods and tools, available at our disposal, based on traditional linear and reductionistic approaches.

We may be able to achieve some success with the above; however, it will only be partial, because linear and reductionistic approaches only study the system components in parts and attempt to estimate important statistics, whereas hydrologic systems are governed by nonlinear feedbacks, bifurcations, chaos, fractals, and emergent properties that run across the individual component parts and require a nonlinear and holistic approach(es).

As much as the concept of a nonlinear and holistic approach is exciting, the following point is noteworthy. Although a nonlinear and holistic approach may be *the* approach for the system as a whole, it may *not* be the one for the individual component parts after all, for which a linear and reductionistic approach may perform better. As a “complete” explanation of a system requires accurate analysis of the parts (reductionism) AND the whole system complexity (holism), a coupled reductionistic-holistic approach may work, in all probability, better than either of the two approaches when adopted independently. The impending challenge, therefore, is to devise such a coupled approach that would provide equal importance to the whole system as well as to its parts.

New theoretical developments and methodological advancements may indeed be needed to address this challenge. To what extent will we give priority to this challenge and how quickly will we be successful in formulating a reliable coupled reductionistic-holistic approach remain to be seen. Although we must start laying the foundation for this task, sooner rather than later, we must also find ways to understand, use, interpret, and even integrate the existing theories and methods better than we have thus far, because the latter may provide some important information that may, subsequently, become significant in implementing the former.

An examination of hydrologic literature clearly reveals the opportunities and possibilities in this regard, as evident from the following two examples. Sivakumar (12), through a discussion of three fundamental ideas of chaos theory (maybe less popular among hydrologists), presents

an interpretation of “chaos theory as a bridge between our popular views of determinism and stochasticity”. Addressing the Dominant Processes Concept, Sivakumar (5) proposes an integration of ideas from nonlinear dynamic theory, expert system, and optimization technique, whose usefulness may be viewed at a much broader spectrum of model simplification.

The fact that interpretation and integration of existing theories and development of new ones often require knowledge of not only hydrology but also other fields, including atmospheric and ocean sciences, ecology, geomorphology, mathematics, and physics, our ability to succeed in identifying and quantifying thresholds mainly lies in our ability to perform collaborative work with researchers in these fields. This topic is discussed next.

INTERDISCIPLINARY RESEARCH ON THRESHOLDS

Thresholds are problem-dependent. Although a connection between thresholds of inputs (e.g., rainfall) and outputs (e.g., runoff), often exists a threshold for an input does not automatically lead to a corresponding threshold in an output. For example, in a rainfall-runoff-sediment transport scenario, the threshold for erosion (a result of runoff) may be different from that for runoff (a result of rainfall), which, in turn, may be different from that for rainfall (a result of atmospheric mechanisms). Similarly, thresholds are also scale-dependent. Although scale-invariance in space and time exists in hydrologic systems and phenomena, different thresholds may exist at different scales, depending on the dominant catchment parameter at the given scale (e.g., slope, land use) and the dominant catchment mechanism (e.g., erosion, interception, evapotranspiration).

These observations indicate the complications involved in even arriving at a definition for thresholds for a given hydrologic system. Also, hydrologic systems play different roles with respect to other related systems (e.g., atmospheric, ecologic, environmental, and geomorphic systems) and vice-versa, which makes the problem even more difficult. Further complications develop when consideration of a one-to-one system scenario or a multiple-system scenario is needed. In view of these, the only hope we can have is to approach the problem from an interdisciplinary view, and multidisciplinary to be more specific, involving researchers from as many related fields as possible.

Establishing interdisciplinary collaborations is not an easy task, particularly at a time we, hydrologists, ourselves seem to be moving away from each other, with our emphasis on specific data analysis techniques rather than common hydrologic problems (6). There is encouraging news, however, as efforts are already being made to address the threshold problem from an interdisciplinary perspective. For instance, the Environmental Hydrosience Discipline Group of the University of Western Australia is working toward the organization of a multidisciplinary workshop entitled “Thresholds and Pattern Dynamics—A New Paradigm for Predicting Climatic Driven Processes,” which is planned to be held July 4–7, 2005 (13). The purpose is to bring together researchers from different disciplines, including seismology, hydrology, ecology, geomorphology, and economics. A session on

introduction of threshold concepts in these disciplines is also planned, an important focus of which is “learning each other’s problems and language,” a point also highlighted by Sivakumar (6) in his proposal of “thresholds” for its simplicity and generality. We will need to make continued efforts in this direction to be successful in solving the threshold “conundrum.”

CONCLUDING REMARKS

Following up on a recent initiation for “thresholds” in hydrology (6), this article discussed their relevance, identification and quantification, and the importance of interdisciplinary research. The discussion points to the need for a new paradigm, or at least a major paradigm shift, for studying thresholds. As thresholds play crucial roles in future predictions and risk assessment, for both the whole system and its component parts, only a (new) coupled reductionistic-holistic approach seems appropriate. Formulation of such an approach, however, may benefit from existing theories and methods, particularly their integration. Interdisciplinary efforts would significantly help in these directions. Hope certainly exists with the availability and advancement of technology, communication facilities, and human resources.

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GANGA RIVER, INDIA

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The Himalayas, the great mountain chain of Asia are the source of three major river systems of the world, the Indus, the Ganga (or the Ganges), and the Brahmaputra. The large and fertile plains of the Indus and Ganga in the subcontinent of Northern India have been the cradles of one of the greatest ancient civilization, the Indus Valley Civilization. The River Ganga which occupies nearly one-third of the geographical area of India is the most important and sacred river of India.

So legendary has been the socioeconomic, cultural, and religious saga of this great river that the Indian mythology and history are full of stories and incidents woven around the river and numerous pilgrim spots along its course like Rishikesh, Hardwar, Allahabad, Varanasi, and Patna.

DESCRIPTION OF GANGA BASIN

The catchment area of the Ganga falls in four countries, India, Nepal, Tibet (China), and Bangladesh. Many important tributaries of the Ganga originate in the Himalayas in India and Nepal; Bangladesh lies in the deltaic region. The total length of the river is 2525 km which makes it the fifteenth longest river in Asia and the thirtieth longest in the world. The combined basin of the Ganga, Brahmaputra, and Meghna rivers in India, Nepal, and Bangladesh is also known by the name *Greater Ganga Basin*. The major part of the geographical area of Ganga Basin lies in India.

Though the headwaters region of the Ganga in the Himalayas is dotted by a number of mighty tributaries, the Bhagirathi river that rises from the Gangotri glacier near Gomukh at an elevation of about 7010 m above mean sea level is traditionally considered the source of River Ganga.

The other main stream that originates in the Uttranchal state of India is the Alakhnanda. Flowing downhill, Bhagirathi and Alakhnanda are joined by a number of streams such as the Mandakini, the Dhuli Ganga, and the Pindar. These two rivers (Bhagirathi and Alakhnanda) meet at a place called Devprayag and thereafter, the combined flow is known by the name Ganga. The River Ganga enters into plains at the pilgrimage center at Haridwar, and from there it flows in a south/southeasterly direction. Yamuna is the most important tributary of the Ganga that joins it on the right bank at Allahabad. The total length of the Yamuna from its origin to Allahabad is 1,376 km, and the drainage area is 366,223 sq. km. The Yamuna is a mighty river in itself and has a number of tributaries such as the Chambal, the Sind, the Betwa, the Ken, and the Tons. After confluence with Yamuna, the Ganga River flows east and is joined by a number of tributaries such as the Ramganga, the Gomti, the Ghaghra, the Gandak, the Bagmati, the Kosi, the Sone and the Damodar.

The Ganga delta is said to begin at a place known as Farakka where a barrage has been constructed to control

river flow. At about 40 km downstream from Farakka, the river splits into two arms. The right arm called River Bhagirathi flows south and enters the Bay of Bengal about 150 km downstream from Calcutta. The left arm, known as Padma, turns east and enters Bangladesh. While flowing in Bangladesh, the Padma meets the River Brahmaputra at a place known as Goalundo. The combined flow, still known as Padma, is joined by another mighty river Meghna at Chandpur, 105 km downstream from Goalundo. Further down, the river ultimately flows into the Bay of Bengal. The delta of the Greater Ganga Basin, one of the largest in the world, is known by the name Sunderbans.

For hydrologic studies, the entire stretch of the Ganga River in India can be divided into three stretches or reaches. The upper reach extends from the origin to Narora, the middle reach from Narora to Ballia, and the lower reach from Ballia to its delta.

An index map of the basin is given in Fig. 1.

HYDROLOGY OF THE GANGA BASIN

The Ganga Basin extends across an area of 1,086,000 sq. km. The drainage area of the basin lying in India is 861,452 sq. km, nearly 26.2% of the geographical area of the country and 79.32% of the Ganga Basin. Some tributaries such as the Ghagra, the Gandak, and the Kosi drain areas in Nepal amounting to 190,000 sq. km. The delta of the Greater Ganga Basin covers an area of 56,700 sq. km. The headwater reaches of the river receive a considerable part of the precipitation as snow, and some peaks are permanently snow covered. The average annual rainfall in the basin varies from 35 cm at the western end to nearly 200 cm near the delta.

The average annual discharge of the Ganga, the Brahmaputra, and the Meghna rivers is 16.65, 19.82,

and 5.1 thousand m^3/s , respectively. The average annual flow of the Ganga at Farraka is about $525 \times 10^9 \text{ m}^3$. There are large variations in the flow of the Ganga with time. Snow and glacier melt during the hot months (March to June) give large summer flows in Himalayan rivers such as the Ganga and its tributaries. The maximum discharge in these rivers is observed during monsoon months (June to September). At Goalundo, the average annual flow of the river is $11,470 \text{ m}^3/\text{s}$. The maximum and minimum flow at this site is 70,934 and $1161 \text{ m}^3/\text{s}$ (1). The peak flow at Farakka in 1971 was estimated at $70,500 \text{ m}^3/\text{s}$.

The surface water resource potential of the Ganga and its tributaries in India has been assessed at $525 \times 10^9 \text{ m}^3$ out of which $250 \times 10^9 \text{ m}^3$ is considered useable NCIWRD (1). Based on the 1991 census, the per capita water availability was near 1471 m^3 per year. Though the Ganga Basin is bestowed with an abundant water resource, its occurrence/availability both in quantity and quality is not uniformly distributed either spatially or temporally. More than 75% of the annual rainfall occurs in the monsoon months of June to September. As a result, large areas are subjected to floods on one hand and droughts on the other.

The storage potential of the Ganga Basin in India has been identified at $84.46 \times 10^9 \text{ m}^3$. However, till 1995, a total of only $36.8 \times 10^9 \text{ m}^3$ of storage space was created. Water resources development schemes to create storage of $17.06 \times 10^9 \text{ m}^3$ are under construction, and projects to provide another $29.56 \times 10^9 \text{ m}^3$ of storage are in the pipeline. The total replenishable ground water resource of the Ganga Basin is estimated at $171.0 \times 10^9 \text{ m}^3$, of which about $48.6 \times 10^9 \text{ m}^3$ was being used by 1999. The River Ganga carries one of the world's highest sediment loads, equal to nearly 1451 million metric tons per annum.

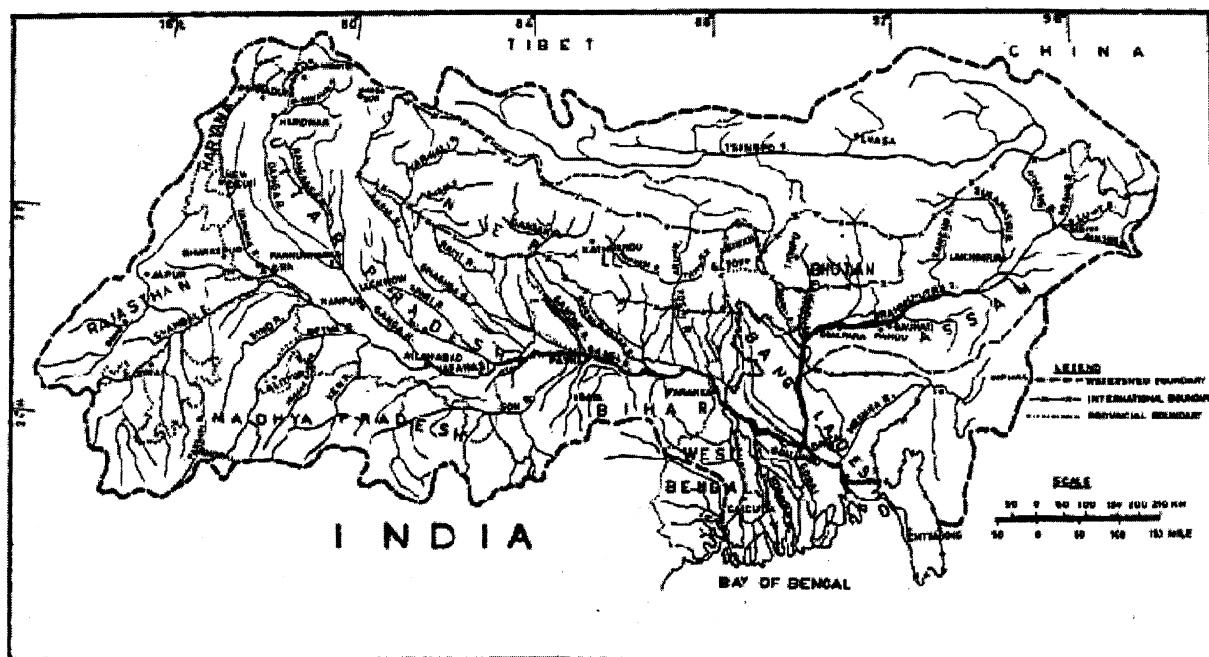


Figure 1. Index map of the Greater Ganga Basin (2).

The Ganga Basin is the largest basin in India, both catchmentwise and populationwise. The total population in the Basin per the 1991 census has been estimated at 356.8 million, which is 42% of India as a whole. The average population density in the Basin was 414 persons per sq. km as against 267 for all of India. Per the 1991 census, the Basin had 111 urban centers that had populations of more than 0.1 million, and of these, seven cities had populations exceeding a million.

The Ganga and its tributaries have formed a large flat and fertile plain in North India. The availability of abundant water resources, fertile soil, and a suitable climate have given rise to a highly developed agriculture-based civilization and one of the most densely populated regions of the world. The net sown area in the Ganga Basin in India is around 44 million hectares (M-ha), and the net irrigated area is 23.41 M-ha.

The hydroelectric potential of the Ganga Basin has been assessed as 10,715 MW at a 60% load factor. Of the 142 identified schemes in the Basin, 22 schemes that have a total installed capacity of 2437 MW are operating and 12 schemes whose installed capacity is about 2716 MW are in various stages of construction.

WATER RESOURCES DEVELOPMENT PROJECTS IN GANGA BASIN

The development of water resource projects in the Ganga Basin has a long history. One of the most remarkable of these works which includes many great civil engineering structures is the Upper Ganga Canal system whose construction was commenced by the then British government in the 1840s to ward off drought and famine in the western part of Uttar Pradesh state. Completed in 1854, the Upper Ganga Canal was to irrigate 0.7 million ha area. The canal takes off from the Ganga at Bhimgoda weir near Hardwar. The main canal is 230 km long and is one of the most exquisitely constructed structures. To expand the command area, the discharge capacity of the canal was recently augmented to nearly 297 m³/s. The command area of the canal is about 0.924 million hectares.

The Madhya Ganga Canal takes off from Ganga at Raoli barrage near Bijnor. The capacity at the head of this 115 km long canal is 234 m³/s. It is operated to provide irrigation to paddy crops in 114,000 hectares as well as to augment the supply to the Upper Ganga Canal system. The Lower Ganga Canal takes off from the Narora weir across the Ganga and was constructed in 1879. The main canal is 100 km long and irrigates 0.5 million ha. Near the Indo-Bangladesh border, a barrage has been constructed at Farakka to divert water to the river Hooghly to increase and maintain adequate depths of flow for navigation and operation of the Calcutta port.

The Yamuna river emerges from the hills near Tajewala where the water is taken off by the western and eastern Yamuna canals. It flows a further 280 km down to Okhla near Delhi from where the Agra Canal takes off. Chambal river is an important tributary of the Yamuna whose catchment area is 139,468 sq. km. Three important storages constructed in the basin include Gandhisagar, Jawahar sagar, and Rana Pratap Sagar cascade, which

provide live storage of 8,500 million cubic m. The barrage at Kota diverts water into canals on either side that irrigate a 0.57 million ha agricultural area.

The Ramganga dam on the tributary of the same name has created a reservoir at Kalagarh that has a live storage capacity of 2,190 million m³. A feeder channel takes off from the dam to provide extra supplies of water to the upper and lower Ganga canal and the Agra canal, besides direct irrigation. A 666,000 ha cropped area will be irrigated by this project. A system of reservoirs in the Damodar valley, Konar, Tilaiya, Tenughat, Maithon, and Panchet dam and Dugapur barrage have largely controlled floods in this subbasin. The Damodar Valley Corporation is responsible for the integrated regulation of most of these dams.

Some other important projects on the tributaries of the Ganga include Matatila on Betwa, Sarda sagar on the Sarda, Obra and Rihand on the Rihand river, and Mayurakshi on the Mayurakshi River. The Sarda canal provides protective irrigation to a nearly 0.6 million ha area. The Tehri dam on Bhagirathi (under construction) will provide live storage capacity of 2,613 million cubic m to be used for power generation and irrigation. Some other major projects under construction are Rajghat on the Betwa, Bansagar on the Sone, and Lakhwar-Vyasi on the Yamuna. A few projects are also being planned in collaboration with the government of Nepal. To boost the use of waterways, some segments of the Ganga River have been declared national waterways.

PROBLEMS IN WATER RESOURCE DEVELOPMENT OF THE GANGA BASIN

The basic problem in using water resources in the Ganga Basin is that in relation to the relatively large annual flow in the Basin, the storage capacity of existing and foreseeable reservoirs in India is not large enough to permit conservation of flows during the high flow season. The live storage capacity of all reservoirs in the Ganga Basin is less than one-sixth of the annual flow, which does not permit the desired degree of flow regulation. The majority of the good storage sites of the Basin are in Nepal, and this is leading to a delay in constructing dams. Lean season flows in the basin without adequate storage backup are not sufficient to meet the requirements for various demands. Besides, monsoon flows in the basin are so high that the Ganga and its tributaries remain in spate almost every year.

In the Ganga Basin, the flooding problem is confined mainly to the middle and terminal reaches. In general, the severity of the problem increases from west to east and from south to north. The worst flood-affected states in the Ganga Basin are Uttar Pradesh, Bihar, and West Bengal. In Uttar Pradesh, flooding is largely confined to the eastern districts; the rivers that cause flooding include the Sarada, the Ghagra, the Rapti, and the Gandak. The major causes of flooding here are drainage congestion and bank erosion. North Bihar is in the grip of floods almost every year due to spillage of rivers. In West Bengal, floods are caused by drainage problems as well as tidal effects. The Ganga

Flood Control Commission was set up by the government of India for flood management in the Ganga Basin.

WATER QUALITY CONSIDERATIONS

The numerous cities located in the Ganga Basin generate and discharge huge quantities of wastewater; a large portion eventually reaches the river through the natural drainage system. Industrial complexes which have sprung up along the rivers also dump considerable pollution loads into the river. Over the years, the Ganga and its tributaries have become the channels of transport of industrial effluents and the drains for the wastewater of cities. It is estimated that some 900 million liters of sewage is dumped into the Ganga every day; three-fourths of the pollution in the Ganga is from untreated municipal sewage. In particular, the middle reach of the basin between Kanpur and Buxar is the most urbanized and industrialized and also the most polluted segment of the Basin. Municipal and industrial wastes in dangerous concentrations enter the watercourse in this segment and pose a grave threat. Murti et al. (3) repeat results of a detailed study on Ganga River.

The pollution load dumped in the river by humans is a serious health hazard to the dense population of the Basin. Recognizing the magnitude of this problem and realizing the importance of water quality as a cardinal element of river management, the government of India started the planning and execution of several programs to check the pollution of the River Ganga and its tributaries. An ambitious program, known as the Ganga Action Plan (GAP), was initiated in 1985. Pollution abatement works for the River Ganga had been taken up in 25 class I towns (population above 1,00,000) along the main River Ganga under the three basin states of Uttar Pradesh, Bihar, and West Bengal. The main objectives of the GAP were

1. reduction of the pollution load on the river and improving the water quality as a result thereof, and
2. establishment of domestic/municipal wastewater treatment systems emphasizing resource recovery to make such systems self-sustainable as far as possible.

In addition, GAP was to serve as a model to demonstrate the methodology of improving the water quality of the other polluted rivers and waterbodies of the country to their designated best use class. A multipronged approach was adopted to achieve the objectives of the GAP. Similar other plans are in various stages of implementation in some tributaries of the Ganga.

The task of restoring the Ganga to its pristine glory will not be easy. Nevertheless, it will be worth the trouble because the sustainable development of a large geographical area and population crucially depends on it.

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INTERCEPTION

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Interception is a key process in hydrology that is receiving relatively little attention. It is the first process of the hydrological cycle that a raindrop experiences after falling from the sky, and hence, all subsequent processes depend on it. Neglect of the interception process can lead to serious modeling mistakes. Interception is defined as the evaporative process that feeds moisture back to the atmosphere shortly after the rainfall has ceased (within approximately 1 day). It is the evaporation from all surfaces that have been moistened by precipitation, including leaves, soil, mulch, rock, paved, and build-up surfaces. Evaporation from interception can amount to a considerable part of all terrestrial evaporation, depending on the climate and region. The interception process behaves differently at different time scales. Formulas are presented for the computation of interception at short, daily, and monthly time scales.

INTRODUCTION

To hydrologists who focus on the terrestrial part of the hydrological cycle, precipitation is generally seen as the starting point of the water cycle. Although one could start with any other point in the cycle as well, e.g., advection of moisture to the land or evaporation from the oceans, it is common to begin the terrestrial water cycle with the rainfall, probably because rainfall is more easily measurable than are other incoming fluxes. This cycle involves many interconnected processes (fluxes), separation points, feedback loops, and stocks where the water resides over longer or shorter periods of time. Starting from the rainfall, the first process that comes into view is interception. It retains the water before it can continue its path in the water cycle, and it allows for a direct feedback loop to the atmosphere. Depending on the precise definition of this process, interception can be understood to mean different things.

First, it is important to decide whether we consider interception as a stock or a flux, or as the combination of both. If we consider it a stock, then it is the amount of rainfall that can be temporarily stored on the land (with its natural and manmade cover) to be evaporated shortly after (or during) the rainfall event. Here, we actually mean the interception capacity. If we consider interception as a

flux, then it is the evaporation from intercepted water, which we express in millimeters per unit of time. If we consider interception as an integrated process, then it is the sum of the rate of change of intercepted water and its evaporation:

$$I = \frac{dS_I}{dt} + E_I \quad (1)$$

where I is the interception process (mm/d), S_I is the interception storage (mm), and E_I is the evaporation from interception (mm/d). The interception I is the part of the rainfall that is intercepted and, after a short period of time, turns into the evaporative flux E_I . For a time scale in the order of 1 day, it is safe to assume that $I = E_I$.

Next, it is important to define the location of the interception process in the hydrological cycle. The most logical place is between the atmosphere and the first separation point where the rainfall splits into interception, surface runoff, and infiltration (see Fig. 1). The interception stock (S_I) is located at the first separation point. In this definition, the interception process includes evaporation from wet leaves, wet land cover (included manmade structures and roads), wet mulch, wet forest floor, and even wet soil. In short, it is the fast evaporation mechanism that dries moist land cover during and directly after the rain.

A more narrow definition of interception, often used, is the difference between rainfall and "throughfall," the rain that falls through the leaves of, e.g., a tree. In this definition, interception is merely the amount of water retained by the leaves of a tree. This definition is not workable for hydrologists because it leaves evaporation from land surface not catered for. Hydrologists who do use this definition, however, combine wet surface evaporation with all other evaporative fluxes and call it evapotranspiration, which is not really a process, but rather a combination of different evaporation processes with different time scales and characteristics (1).

The total evaporation E in a catchment is the sum of several different processes: interception I , transpiration

T , surface evaporation E_s , and open water evaporation E_o (2). In the broader definition of interception, we combine evaporation from leaf interception with wet surface evaporation occurring on the same day as the rainfall. The amount of interception thus defined is larger than the amounts quoted in the literature based on the difference between rainfall and throughfall.

THE IMPORTANCE OF INTERCEPTION

Interception is one of the most underrated, and underestimated, processes in rainfall–runoff analysis. One could call interception the "Cinderella of Hydrology." She may be beautiful and rather straightforward but apparently not interesting enough to be adequately represented in hydrological models. Some models even disregard interception completely [particularly event-based models, based on the simplified catchment model of Dooge (3, 4)], the argument being that it is generally a small proportion of the total evaporation. This disregard is a mistake, for several reasons.

First, the amount of interception is not small. Beven (5) states that evaporation from intercepted water on leaf surfaces in rough canopies can be efficient and a significant component of the total water balance in some environments. Calder (6) shows that in the upland forest catchments of Britain, evaporation from interception amounts to 35% in areas with an annual rainfall of more than 1000 mm/a, but that it is higher in areas with lower rainfall, amounting to about 40–50% in areas with 500–600 mm/a. In Calder's definition, however, interception is merely the difference between rainfall and throughfall. But if we use the broader definition presented above, interception is considerably larger. After a rainfall event, not only the leaves of vegetation are wet, but also the surface underneath (rock, soil, mulch layer, roads, build-up area, etc.), which becomes dry within the same day, particularly in warm climates. Moreover, many stagnant pools continue to evaporate until a day after the event. The additional wet surface evaporation can be as much as the interception by leaves, particularly in dry climates. These interception processes are fast, having an average residence time in the order of 1 day.

Note that the wetted soil surface should not be considered part of the soil moisture that feeds the transpiration process. The wet surface (extending to several millimetres of soil depth) feeds back the intercepted water through direct evaporation and not via a delayed transpiration process. Even a stretch of dry sand, without vegetation, can intercept water. After a rainfall event, a wet "crust" of soil is formed, underlain by dry sand, which dries out again within a day. This soil can intercept several millimeters of rainfall.

Transpiration is different. First, it is a physiological process intimately tied to CO_2 assimilation. The time scale of transpiration is determined by the soil moisture stock, which makes the time scale of the process much longer (average residence times varying between weeks and months depending on the soil depth). Finally, the process of transpiration does not change the isotope composition of

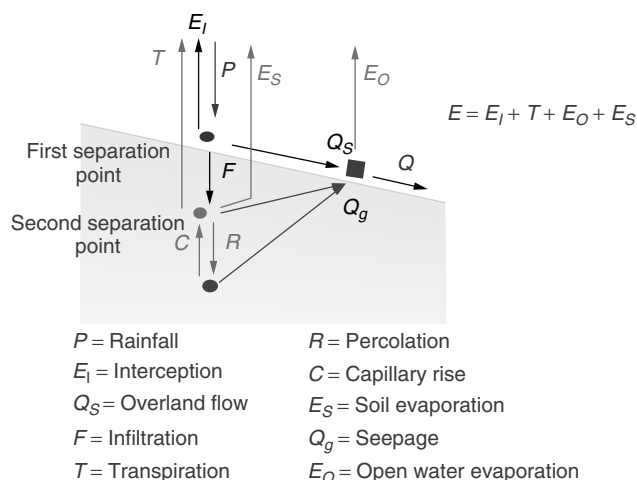


Figure 1. Separation points at the landscape interface.

the evaporated moisture, whereas evaporation of a water surface does. One can distinguish fast transpiration and delayed transpiration. Fast transpiration is from shallow rooted plants (typically grass and annual crops) with a time scale of less than a month; delayed transpiration is from deeply rooted plants (trees, shrubs, perennial crops), which have a time scale longer than a month. Fast transpiration only draws on the upper soil layer (until 50 cm depth), whereas delayed transpiration draws on deeper soil layers. Open water evaporation can be considered separately, when necessary, and is identifiable relatively simply.

DETERMINING EVAPORATION FROM INTERCEPTION

Computation of interception on an instantaneous basis is done by a Rutter model (7), based on Equation 1. For practical purposes, however, and in view of the typical time scale of the process where wetness caused by a rainfall event evaporates within a time span of a day, interception is well captured on a daily time scale. The daily process of interception is conveniently described as a threshold process, with a daily threshold D (mm/d). Daily interception (mm/d) is then computed as

$$I = \min(P, D) \quad (2)$$

which implies that if the rainfall is less than the threshold, the interception is equal to the rainfall and otherwise it is equal to the threshold value.

Monthly evaporation from interception can be determined efficiently, even for poorly gauged catchments through De Groen's method. De Groen (8) developed an analytical model for the computation of monthly interception based on the Markov property of daily rainfall¹ and using a daily interception capacity D .

$$I_m = P_m \left(1 - \exp \left(-\frac{n_r D}{P_m} \right) \right) \quad (3)$$

where I_m is the monthly evaporation from interception (mm/month), P_m is the monthly rainfall (mm/month), and n_r is the number of raindays per month (d/month). She demonstrated that, as a result of the Markov property of daily rainfall, the number of raindays in a month is a function of the monthly rainfall.

$$n_r = \frac{30p_{01}}{1 - p_{11} + p_{01}} \quad (4)$$

where p_{01} is the probability of a rainday after a dry day and p_{11} is the probability of a rainday after a rainday. These probabilities appear to be power functions of monthly rainfall. These power functions can be derived from time series analysis of selected raingauges with daily observations. De Groen (9) demonstrated that the

coefficients of the power functions are representative for larger areas and can be interpolated between rainfall stations, similar to the regional distribution of monthly rainfall, which is surprising, because daily rainfall is known to have a very small spatial correlation (a few kilometer), but apparently the Markov property has a high one, of the same order of magnitude as monthly rainfall (100–1000 km).

Figure 2 shows the relative contributions of monthly interception (I) and transpiration (T) to total evaporation (E) in the Mupfure catchment. The Mupfure river, at Beatrice in Zimbabwe, drains a small catchment of 1215 km². It has an annual rainfall of about 800 mm/year. There is no substantial open water. The plots are based on monthly values over the period 1970–1979. The monthly interception has been computed by De Groen's formula with a threshold value of 4 mm/day. This threshold, may seem large but it caters to more than the difference between rainfall and throughfall (including wet surface evaporation) and there is generally more than one rainfall event per day, each of which contributes to evaporation from interception. Pitman (10) argues that interception in Southern Africa can be as much as 8 mm/day in forests. De Groen (8) considers 2–5 mm/day appropriate, depending on the land use, but the true value is not exactly known. The use of a higher or lower value of D changes the partitioning between I and T . The transpiration is computed as the difference between the interception and the total evaporation, which was determined on the basis of a monthly water balance. As a result, Fig. 2 gives an impression of how interception and transpiration relate and their order of magnitude, for different amounts of monthly rainfall.

As an aside, in Fig. 2, one can distinguish scatter dots and dots that appear to follow a pattern (the drawn lines). The scatter dots that occur in the low rainfall months are the result of transpiration from soil moisture stocks built-up during the wet season, which are accessed by deeply rooted vegetation. The scatter dots in the months with high rainfall are caused by limited solar radiation constraining evaporation. The drawn curves represent the contributions of interception and transpiration to the fast evaporation, occurring within 1 month, i.e., interception and fast transpiration.

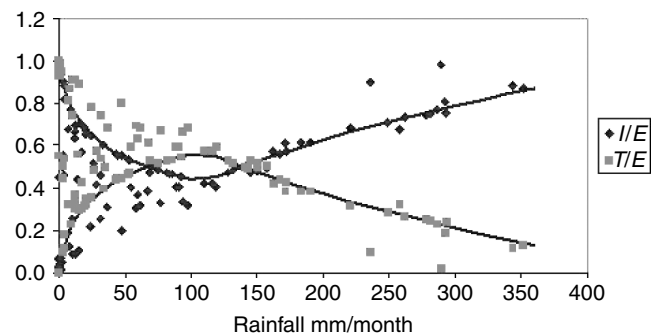


Figure 2. Relative contributions of interception (I) and transpiration (T) to monthly evaporation (E) in the Mupfure catchment in Zimbabwe, with $D_d = 4$ mm/day.

¹The Markov property implies that the probability that a certain day is a rainday depends purely on whether the previous day was a wet or a dry day.

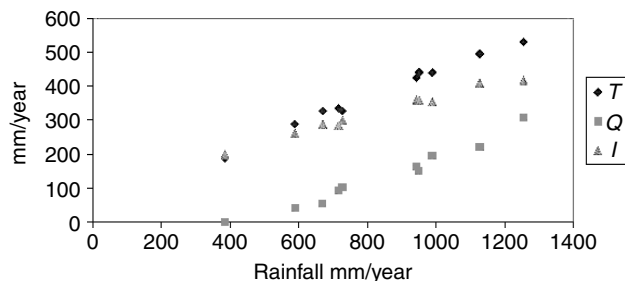


Figure 3. Annual rainfall partitioning into runoff (Q), interception (I), and transpiration (T), Mupfure basin at Beatrice, with $D_d = 4$ mm/day.

It can be seen that, overall in the Mupfure catchment, evaporation from interception is as significant as transpiration, and that in the wet months, transpiration is dominated by fast transpiration (the drawn curve), as there is no moisture deficit and the soil moisture stock is being replenished. More importantly, interception is dominant in wet months. Fig. 3 shows the annual partitioning of rainfall into transpiration (T), runoff (Q), and interception (I), indicating that interception comprises a considerable amount of the annual rainfall, with a tendency for interception to dominate in very dry years.

Besides interception forming a considerable part of monthly evaporation, it is crucial for determining antecedent conditions in rainfall–runoff modeling. It is often said that during storm events, evaporation from interception is negligible as compared with other fluxes. This may be true during the event, which is of short duration, but it is untrue in relation to the build-up of the antecedent conditions. It is widely recognized that the success of event-based modeling depends to a large extent on the antecedent conditions, particularly the distribution of the soil moisture in the unsaturated soil. If we want to get that right, it is important to split the rainfall into the effective part that contributes to soil moisture (and hence the rainfall–runoff process) and the ineffective part that does not, i.e., interception.

Finally, evaporation from interception is the most important process in moisture recycling to support continental rainfall. Figure 2 illustrates this. It shows that in months with high rainfall (i.e., the wet season), interception is an important mechanism. In dry months, transpiration is often dominant, but in these months (typically after the wet season), there are hardly any rainfall generating mechanisms to benefit from this feedback. Shuttleworth (2) observed that half the evaporation from interception occurs during the storm, which provides instant moisture feedback. Hence the moisture feedback to the atmosphere, which is such an important mechanism to support continental rainfall in the Sahel, and the Amazon (11,12), relies primarily on interception. The reason why transpiration is relatively small in wet months is because part of the energy available for evaporation is consumed by the interception process, which precedes transpiration, and because solar radiation is inhibited by clouds in wet months.

CONSEQUENCES OF UNDERESTIMATING INTERCEPTION

The picture of the Mupfure basin may not be representative for other climatic regions, but it serves to make a point. Interception is an important mechanism that cannot be simply neglected or lumped with other evaporation mechanisms. Disregarding or underestimating interception can lead to serious modeling mistakes, particularly when one uses automated calibration techniques. If interception is modeled incorrectly, the error will be compensated by other parameters, to satisfy the goodness-of-fit criterion. If a range of interception thresholds is tested, then one may find a correlation between interception and other soil and groundwater parameters. Clearly such relations are spurious.

The most common mistake of lumping interception with transpiration leads to an overdimensioning of the soil moisture stock, which can be seen easily. If the interception is forced through the transpiration process, a correct representation of the total flux and time scale in the model can only be achieved if the soil moisture stock is overdimensioned. If transpiration and interception are of the same order of magnitude, then the modeled soil moisture stock should be double its “real” value.

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KINEMATIC SHOCK

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Kinematic shocks form due to a variety of factors, including spatial and temporal variability of inflow, systems geometry, and initial and boundary conditions. This study provides a short overview of the formation, propagation, tracing, and fitting of shocks in hydrologic modeling, in particular, referring to rainfall–runoff modeling.

INTRODUCTION

The kinematic shock is a discontinuity representing a sudden rise or surge in flow depth. For example, flood waves have an intrinsic tendency to steepen as they propagate downstream due to lateral inflow or geometric constriction and eventually form a shock. The discontinuity can also represent a sudden decrease in flow depth. For example, when flow advances over a porous bed, there comes a time when the flow rate is not sufficient to satisfy the infiltrating demand of the bed and is, therefore, suddenly absorbed by the bed. This phenomenon is observed in ephemeral streams and irrigation borders and furrows. Discontinuities in flow are also observed in geologic formations due to fracture, fissures, and sudden changes in porous media properties. Shocks are also observed in landscape evolution and formation of deltas, river bed profiles, and river junctions.

Documented evidence of the occurrence of shocks is hard to find, but shocks have been observed in numerical solutions (1). Shocks lead to physically unrealistic multiple-valued and nonconvergent solutions (2). Therefore, proper consideration of shocks in hydrologic and hydraulic modeling is needed. There are four aspects of shocks that need consideration: (1) formation, (2) propagation, (3) tracing of the shock path, and (4) shock fitting. From a numerical standpoint, two main issues need to be addressed: (1) the physical relevance of kinematic shocks and (2) the adequacy of numerical schemes in the presence of shocks. In many kinematic cascade models, numerical schemes, such as the Lax–Wendroff scheme, do not exhibit multiple-valued solutions, and therefore, the presence of shocks goes unnoticed. This may partly be due to smoothing out of shocks by the numerical schemes. For many engineering problems, this may be acceptable if shocks are small and of short duration. However, these schemes may yield unrealistic or undesirable multiple hydrographs. Then, kinematic shocks warrant proper consideration.

SHOCK FORMATION

A range of factors leads to shock formation. These factors can be classified into three groups. In the first group are initial and boundary conditions. Lateral inflow and outflow form the second group. System geometric

characteristics, such as surface roughness, geometric shape, and surface slope, are in the third group. It is, however, difficult to single out the contribution of each individual factor. Ponce and Windingland (3) attempted to determine flow and channel characteristics that either tend to promote or inhibit development of a kinematic shock. The flow and channel characteristics considered by them were the inflow hydrograph peak, the Froude number, the time to peak, the base-to-peak-flow ratio, the kinematic roughness parameter, and the kinematic exponent. The inflow hydrograph was represented by a gamma distribution. Of all the parameters, the size of the wave was the most important in the occurrence or nonoccurrence of the shock. They concluded that a kinematic shock is most likely to occur when there are a low base-to-peak-flow ratio, a high Froude number, and a wide and sufficiently long channel. A flood wave traveling in a steep, initially dry channel would be a potential candidate for shock formation.

ORDER OF DISCONTINUITY

Two kinds of discontinuities occur in hydrology. A discontinuity of the first kind (or first order) represents a sudden, finite change in flow depth. In that case, the differential equation of continuity no longer applies. The partial derivatives cease to be defined so that some additional condition is needed. This condition may be derived using the theory of weak solutions replacing the differential equation by an integral equation or by multiplying the conservation of mass to a frame comprising the shock (4). Both methods yield the same result in the form of a jump condition. According to the second method, the differential continuity equation is replaced by an equation stating that the flow into one side of the shock equals the flow out on the other side. Mathematically, this can be expressed as

$$U = \frac{Q_2 - Q_1}{h_2 - h_1} = \frac{\Delta Q}{\Delta h} \quad (1)$$

where subscripts 1 and 2 denote the variables calculated in front of and behind the moving shock, U is the shock velocity which must be between the characteristic velocities on either side of the shock (5), h is the flow depth, and Q is the flow discharge.

A discontinuity of the second kind (or second order) is a small change in flow depth h . If $h_2 - h_1 = dh$ is small, the expression for U reduces to

$$U = \frac{dQ}{dh} = c(h) \quad (2)$$

where c is the velocity of the discontinuity between flow depths h and $h + dh$. A small change in flow depth, dh , if maintained, is propagated through a flow depth h at velocity U . A line of constant depth in the $x-t$ diagram, therefore, describes the motion of the boundary between flows of depth h and $h + dh$, such that its slope is necessarily equal to the flow wave celerity, c .

For flows where the depth (h) increases downstream, the condition for forming a discontinuity of the first kind

can be expressed in the following equivalent terms: (1) The lines of constant depth in the $x-t$ diagram, if continued away from the t axis or x axis, would intersect; (2) the wave celerity, c , increases with depth of flow; and (3) the $Q-h$ curve is concave to the h axis. If these conditions are not satisfied, a first-order discontinuity is not formed. This can be illustrated in terms of second-order discontinuity. If c increases with h , small depth changes in higher flow regions upstream will move faster downstream than those in lower order flow regions and overtake them. This means that the depth gradient increases until a first-order discontinuity is formed. If c decreases with h , the reverse takes place; the depth gradient decreases, and any discontinuity is dispersed.

Three types of first-order shocks can be identified. To that end, we consider an $x-t$ diagram which has two types of characteristics: A and B, as shown in Fig. 1. The A characteristics originate from the x axis and therefore depend on the initial condition. The B characteristics originate from the t axis and therefore depend on the boundary condition. These two types of characteristics are separated by a bounding or limiting characteristic that originates from the origin. The first kind of shock (or discontinuity) forms when two A characteristics intersect. This kind of shock does not occur in overland flow on an initially dry plane where effective rainfall depends only on time. The second kind of shock is designated as mixed shock and is formed by the intersection of one A characteristic and one B characteristic. Clearly, the limiting characteristic plays a major role in this case. The

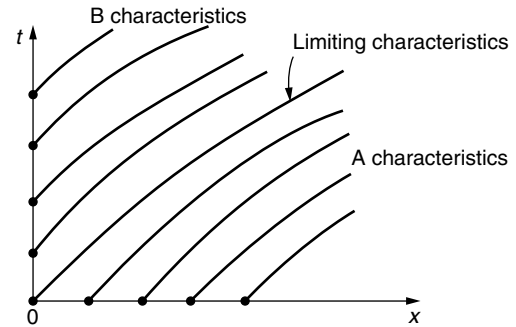


Figure 1. Characteristics of a plane in the (x, t) space.

third kind of shock is boundary dependent and forms when two characteristics intersect.

It may be interesting to consider some possible cases of boundary-dependent shock formation. When rainfall increases with time and c increases with h , a shock will be formed by the intersection of the B characteristics, as shown in Fig. 2a. This can be prevented only by stopping it at the discontinuity. The shock is small and is fed by the characteristics running into it. Another example is when the initial region of varying h is small and there is effectively an initial discontinuity that is propagated along the line of discontinuity, as shown in Fig. 2b. Again, the discontinuity is small and is fed by lines running into it.

Another case is a sudden rise in rainfall at a given time within a small region of rapid change. A shock is

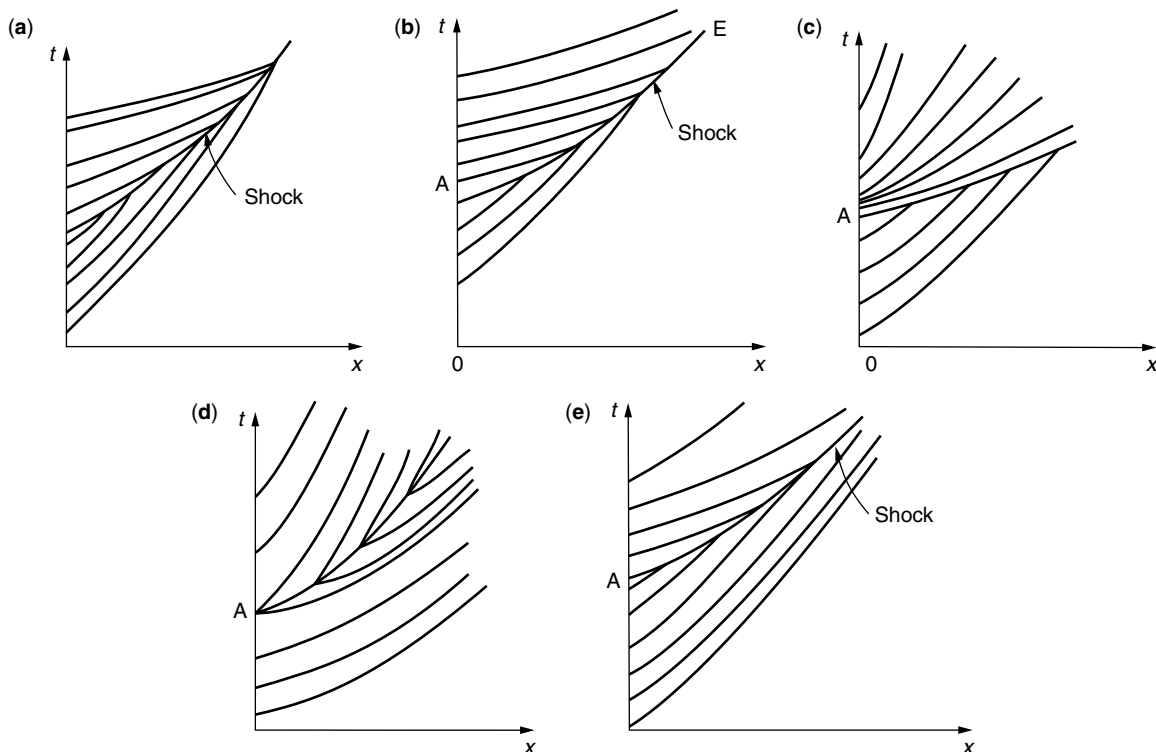


Figure 2. (a) Rainfall increases continuously with t ; (b) rainfall increases over a short time; (c) rainfall suddenly increases at A in the limit; (d) rainfall increases, but without limit; and (e) rainfall increases to a limit and then decreases.

formed and occupies a region. It is fed by characteristics running into its lower boundary, as shown in Fig. 2c. This assumption is waived in Fig. 2d. Still another case occurs when rainfall increases to a limit and then decreases. In this case also, the shock occupies a region, as shown in Fig. 2e.

SHOCKS IN PLANAR FLOWS

For constant effective rainfall, r , the B characteristics are given by (6)

$$h(t, t_0) = h_1(t_0) + r(t - t_0) \quad (3)$$

$$x(t, t_0) = \frac{\alpha}{r} [h_1(t_0) + r(t - t_0)]^n - \frac{\alpha}{r} h_1^n(t_0) \quad (4)$$

where $t_0 (0 \leq t_0 \leq \infty)$ is the time when a B characteristic intersects the t axis, h_1 is the flow depth at the boundary $x = 0$, n is the kinematic flow exponent, and α is the kinematic roughness coefficient. A shock may occur if $\partial x(t, t_0) / \partial t_0 > 0$, and the derivative can be obtained by differentiating Eq. 4. The region of intersecting characteristics can be obtained by taking the derivative of Eq. 4 with respect to t_0 and equating them to zero. Then, these two derivative equations can be solved to yield the coordinates of points on the characteristic envelope. The smallest values of these coordinates define the first point in the region of B characteristics where the formation of the kinematic shock is initiated, as shown in Fig. 3.

SHOCKS IN A KINEMATIC CASCADE

The mathematical properties of a kinematic cascade were investigated by Kibler and Woolhiser (1,7). A cascade is a representation of a watershed or a portion thereof by a series of planes discharging into a channel. Thus, the outflow of one plane forms the upper boundary condition of flow for the next. They developed a criterion, based on the properties of adjacent flow-plane pairs, to predict when a shock occurs in a cascade. They modified their Lax–Wendroff solution by placing instantaneous jumps in the output hydrograph corresponding to the arrival of shocks. They found that the Lax–Wendroff scheme effectively smoothed out shocks but was adequate when

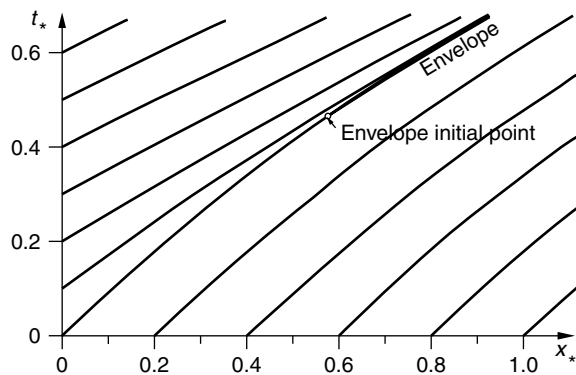


Figure 3. Intersection of characteristics in the (x, t) plane.

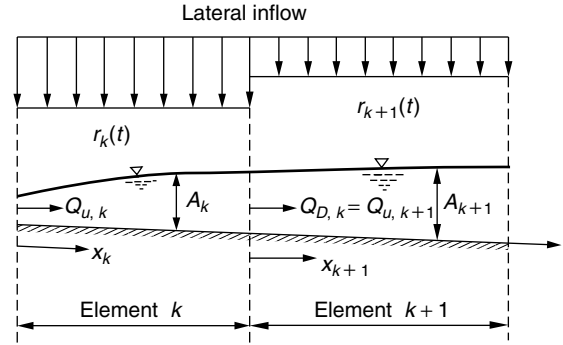


Figure 4. Two geometric elements in series, each has spatially uniform lateral inflow and topographic characteristics.

corrected for known shocks for solving the kinematic cascade. Schmid (8) derived a generalized shock formation criterion for infiltrating cascades that also encompasses the cascades with time-dependent rates of effective rainfall. On an infiltrating surface, the time to ponding marks the onset of overland flow. Ponding time can be computed by using the hydraulic properties of soils, and it strongly influences shock formation. For an infiltrating cascade, the ratio of respective ponding times of two adjacent, infiltrating planes is of major importance.

In a kinematic cascade, the criterion for shock formation is when the time to ponding of the upper plane is less than the time to ponding of the lower plane. If a cascade has two planes and overland flow starts on the upper plane rather than on the lower plane, a shock will form as soon as the rate of effective rainfall on the upper plane becomes nonzero, as shown in Fig. 4. Shocks can also occur when the time to ponding on the upper plane is greater than that on the lower plane (6).

SHOCKS ON CURVED SURFACES

A curved surface, where the slope continuously varies, can be considered a generalization of the kinematic cascade. Harsine and Parlange (9) dealt with kinematic shocks on curved surfaces. If the kinematic roughness coefficient varies in space or is a function of x , then one condition for the shock to occur is when $d\alpha_0/dx_0$ is a maximum. When $x = x_0$, the points where the B characteristics originate from the x axis, at that point α is α_0 . Similarly, the effect of spatial variations on other surface geometric features can be derived.

SHOCK FITTING

Due to the spatial and temporal variability of rainfall and watershed characteristics, innumerable shocks may develop when the kinematic wave theory is applied to rainfall–runoff modeling, flow through unsaturated media, flow over porous beds, and dam break modeling. Because shock formation is intrinsic to the theory, approximate methods of shock fitting have been developed and are employed. In general, shock fitting entails (1) determining the characteristics intersecting each other

on the shock path, (2) locating the points of intersection or determining the shock path, and (3) shock fitting.

It is assumed that a shock or discontinuity in h occurs first at the initial point on the characteristic envelope, and then the location of this moving shock is traced in the $x-t$ plane as the solution of the kinematic wave equations is generated. Across this discontinuity, however, the differential form of the continuity equation does not hold, but a control-volume form of this equation does, which leads to Eq. 1. In this equation, discharge is expressed by the kinematic depth–discharge relation. Thus,

$$\frac{dx_s}{dt} = \alpha \frac{h_2^n - h_1^n}{h_2 - h_1} \quad (5)$$

where x is the x coordinate of the discontinuity. The path of this moving shock in the $x-t$ plane results from the solution of Eq. 5:

$$x_s(t) = x_0 + \alpha \int_{t_0}^t \frac{h_2^n - h_1^n}{h_2 - h_1} \quad (6)$$

where (x_0, t_0) are the coordinates of the initial point on the envelope. An iterative numerical solution of Eq. 6 is straightforward.

Borah et al. (10) developed an approximate procedure for shock fitting in unsteady-state flow routing. This method avoids finding the exact location of the shock origin by discretizing the upstream hydrograph, and it preserves the effect of shocks by routing them as they appear. It permits the use of the same numerical scheme to route both characteristic and shock waves. The procedure is particularly efficient when the initial condition represents a dry or uniform flow condition and only the outflow hydrograph is desired.

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KINEMATIC WAVE METHOD FOR STORM DRAINAGE DESIGN

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INTRODUCTION

Ever since Lighthill and Whitham showed that the main body of a natural flood wave moves as a kinematic wave, there has been continual interest in applying the kinematic wave theory to drainage design. A drainage design method based on the kinematic wave theory has several inherent advantages: physically based parameters, facility for hydrograph prediction, and feasibility of obtaining analytical solutions for certain idealized basin and channel conditions. In this article, based on the theory, a drainage design method is developed for estimating the design discharge for a series of overland planes and for a V-shaped basin.

APPLICABILITY OF KINEMATIC WAVE THEORY

The applicability of the kinematic wave theory to overland and open channel flow of with sufficient accuracy compared to the solution from the Saint-Venant equations has been investigated by several researchers (1–3). For overland flow, the applicability of the theory can be defined by the Morris and Woolhiser (13) criterion,

$$kF_o \geq 5 \quad (1)$$

where k is the kinematic flow number and F_o is the Froude number at the end of the plane at equilibrium. The parameter kF_o can be related to the physical characteristics of an overland plane and the rainfall intensity as follows (4, p. 66):

$$kF_o = 8586 \frac{S_o^{1.3} L_o^{0.4}}{n_o^{0.6} i_n^{0.6}} \quad (2)$$

where S_o is the slope of the overland plane, L_o is the length of the overland plane, n_o is Manning's roughness coefficient of a plane surface, and i_n is the net rainfall intensity. The units are m m^{-1} for S_o , m for L_o , and mm h^{-1} for i_n .

For open channel flow, the applicability of the theory can be defined by the Ponce et al. (2) criterion,

$$\tau = \frac{TS_c u_c}{y_c} > 83 \quad (3)$$

where τ is the dimensionless wave period, T is the wave period that can be taken as twice the time-of-rise of the flood wave (5), S_c is the channel bed slope, u_c is the steady-state, uniform, channel flow mean velocity, and y_c is the steady-state uniform flow depth in the channel. The units are s for T , m m^{-1} for S_c , m s^{-1} for u_c and m for y_c . As a rule of thumb, the American Society of Civil Engineers (6, p. 58; 7, p. 87) simplified the criterion to

$$S_c > 0.002 \quad (4)$$

In general, the theory is applicable to overland flow and most open channel flow where the backwater effect is not significant (8, p. 42).

RAINFALL INTENSITY–DURATION RELATIONSHIP

To estimate the design discharge of a desired recurrence interval, the rainfall intensity–duration curve of the same recurrence interval is used. Analyses of the total rainfall curves show that, for a given recurrence interval, the rainfall intensity varies inversely with the rainfall duration and can be mathematically described by (9; 10, p. 71)

$$i = a/(c + t_r)^b \quad (5)$$

where i is the rainfall intensity, t_r is the rainfall duration; and a , b , and c are constants. To facilitate the derivation of an explicit expression for design discharge, Eq. 5 is reduced to (11, p. 20)

$$i = at_r^{-b} \quad (6)$$

Using Eq. 6 with a single set of a and b values cannot fit the entire rainfall intensity–duration curve, but Chen and Evans (12) showed that by dividing the rainfall curve into segments, it is possible to fit the entire rainfall curve using different values of a and b for each segment.

Further, the effect of abstraction losses such as initial and infiltration losses can be incorporated into Equation 6 by substituting the net intensity, i_n , for rainfall intensity, i , and the net duration, t_n , for rainfall duration, t_r :

$$i_n = a_n t_n^{-b_n} \quad (7)$$

where a_n and b_n are constant within one segment of the rainfall curve but vary from segment to segment. The relationship between the net intensity, i_n , the rainfall intensity, i , and the rate of infiltration, f , is

$$i_n = i - f \quad (8)$$

and the relationship between the net duration, t_n , the duration, t_r , and the initial loss, I , is

$$t_n = t_r - \left(\frac{60I}{i} \right) \quad (9)$$

The units for Eqs. 6–11 are mm h^{-1} for i , i_n , and f ; min for t_r and t_n ; and mm for I . Typical values of initial loss and uniform infiltration rates for various surface types are reported by Stephenson and Meadows (4, p. 57).

TIME OF CONCENTRATION FOR A SERIES OF PLANES

Based on the kinematic wave theory, Wong (13) showed that the time of concentration, t_o , for a series of planes, is

$$t_o = \sum_{j=1}^N 7L_{oj} \times \left(\frac{n_{oj}}{\sqrt{S_{oj}}} \right)^{3/5} \left\{ \frac{\left[\sum_{r=1}^j (i_{nr} L_{or}) \right]^{3/5} - \left[\sum_{r=1}^{j-1} (i_{nr} L_{or}) \right]^{3/5}}{\sum_{r=1}^j (i_{nr} L_{or}) - \sum_{r=1}^{j-1} (i_{nr} L_{or})} \right\} \quad (10)$$

where N is the number of planes, j is the j th plane under consideration in the direction of flow for calculating the time of concentration, and r is the r th plane under consideration in the direction of flow for calculating the equilibrium inflow and outflow of the j th plane. The units are min for t_o , m for L_o , m m^{-1} for S_o , and mm h^{-1} for i_n . Equation 10 is applicable to a series of planes in which each plane has uniform slope and roughness, is subject to uniform net rainfall intensity, and has a turbulent or nearly turbulent flow regime. Within this condition, it can be applied to a cascade of planes, to planes of different roughness, to planes of different soil types resulting in different net intensities, to planes subject to different rainfall intensities, and to planes that combine all these variables. In the application of Eq. 10, the values of n_o may be selected from the recommended values of Engman (14), the Institution of Engineers, Australia (15, p. 300), or the American Society of Civil Engineers and Water Environment Federation (10, p. 88).

If the net intensity is the same for all planes, Eq. 10 reduces to

$$t_o = \sum_{j=1}^N \frac{7}{i_n^{2/5}} \left(\frac{n_{oj}}{\sqrt{S_{oj}}} \right)^{3/5} \left[\left(\sum_{r=1}^j L_{or} \right)^{3/5} - \left(\sum_{r=1}^{j-1} L_{or} \right)^{3/5} \right] \quad (11)$$

If there is only one plane, both Eqs. 10 and 11 reduce to (1)

$$t_o = \frac{7}{i_n^{2/5}} \left(\frac{n_o L_o}{\sqrt{S_o}} \right)^{3/5} \quad (12)$$

DESIGN DISCHARGE FOR A SERIES OF PLANES

For a homogenous, rectangular plane, Stephenson and Meadows (4, pp. 49–51) and Chen and Wong (16) showed that the maximum peak discharge for the plane may occur under the condition of full-area or partial-area contribution. Chen and Wong (17) and Wong (18) further showed if the maximum discharge occurs under the condition of partial-area contribution, the maximum discharge is related to the rainfall by $b_n = 1$ in Equation 7. However, this condition may not apply to a series of planes that are heterogeneous. A trial and error process is required to determine the maximum discharge for the series of planes. The maximum discharge is the design discharge for the planes.

For planes subject to uniform net rainfall intensity and for full-area contribution, an explicit expression for the peak discharge can be derived by equating the time of concentration of the planes to the net rainfall duration. This expression can also be used to estimate the peak discharge from the downstream portion of the series of planes. For full-area contribution, the peak discharge per unit width of plane, q_o , is related to the design rainfall intensity, i_o , as follows:

$$q_o = \frac{i_o \sum_{j=1}^N L_{oj}}{3.6 \times 10^6} \quad (13)$$

The units are $\text{m}^2 \text{s}^{-1}$ for q_o , mm h^{-1} for i_o , and m for L_o . Equating t_n Eq. 7 to t_o Eq. 11 to obtain the design rainfall

intensity, i_o , and substituting it into Eq. 13 results in the following explicit expression for the peak discharge q_o :

$$q_o = \frac{\sum_{j=1}^N L_{oj}}{3.6 \times 10^6} \left\{ \frac{a_o^{1/b_o}}{\sum_{j=1}^N 7 \left(\frac{n_{oj}}{\sqrt{S_{oj}}} \right)^{3/5}} \right\}^{b_o/[1-(2b_o/5)]} \times \left[\left(\sum_{r=1}^j L_{or} \right)^{3/5} - \left(\sum_{r=1}^{j-1} L_{or} \right)^{3/5} \right] \quad (14)$$

where a_o and b_o are the respective values of a_n and b_n in Eq. 7 for $i_n = i_o$ and $t_n = t_o$. The units are $\text{m}^2 \text{s}^{-1}$ for q_o , m for L_o , and m m^{-1} for S_o .

If there is only one plane, Eq. 14 reduces to

$$q_o = \frac{1}{3.6 \times 10^6} \left\{ \frac{a_o^{1/b_o} L_o^{(1-b_o)/b_o}}{7 \left(\frac{n_o}{\sqrt{S_o}} \right)^{3/5}} \right\}^{b_o/[1-(2b_o/5)]} \quad (15)$$

TIME OF TRAVEL IN A CHANNEL

For a drainage basin that comprises overland planes and drainage channels, the time of concentration for the basin is the summation of the overland time of concentration and the time of travel in the channel (8, p. 90). Yen (19) asserted that these two flow components should be evaluated separately, as they are really two separate, sequential systems. Further, Kibler and Aron (20) showed that time of concentration methods that consider the two flow components separately give better estimates.

Based on the kinematic wave theory, Wong (21) and Wong and Zhou (22) showed that for channels that have a negligible backwater effect subject to uniform lateral inflow and constant upstream inflow, the times of travel in these channels for seven different cross sections are as follows:

1. Wide rectangular channel

$$t_t = 0.0167 \left(\frac{n_c}{\sqrt{S_c}} \right)^{3/5} L_c B^{2/5} \left(\frac{Q_d^{3/5} - Q_u^{3/5}}{Q_d - Q_u} \right) \quad (16)$$

2. Square channel

$$t_t = 0.0289 \left(\frac{n_c}{\sqrt{S_c}} \right)^{3/4} L_c \left(\frac{Q_d^{3/4} - Q_u^{3/4}}{Q_d - Q_u} \right) \quad (17)$$

3. Deep rectangular channel

$$t_t = 0.0265 \left(\frac{n_c}{\sqrt{S_c}} \right) \frac{L_c}{B^{2/3}} \quad (18)$$

4. Triangular channel

$$t_t = 0.0236 \left(\frac{n_c}{\sqrt{S_c}} \right)^{3/4} L_c \left(\frac{z^2 + 1}{z} \right)^{1/4} \times \left(\frac{Q_d^{3/4} - Q_u^{3/4}}{Q_d - Q_u} \right) \quad (19)$$

5. Vertical curb channel

$$t_t = 0.0198 \left(\frac{n_c}{\sqrt{S_c}} \right)^{3/4} L_c \left(\frac{1 + \sqrt{z^2 + 1}}{\sqrt{z}} \right)^{1/2} \times \left(\frac{Q_d^{3/4} - Q_u^{3/4}}{Q_d - Q_u} \right) \quad (20)$$

6. Parabolic channel

$$t_t = 0.0272 \left(\frac{n_c}{\sqrt{S_c}} \right)^{9/13} L_c B^{2/13} \left(\frac{Q_d^{9/13} - Q_u^{9/13}}{Q_d - Q_u} \right) \quad (21)$$

7. Circular channel

$$t_t = 0.0290 \left(\frac{n_c}{\sqrt{S_c}} \right)^{4/5} \frac{L_c}{B^{2/15}} \left(\frac{Q_d^{4/5} - Q_u^{4/5}}{Q_d - Q_u} \right) \quad (22)$$

where t_t is the time of travel in the channel, n_c is Manning's roughness coefficient of the channel surface, S_c is the channel bed slope, L_c is the channel length, B is the dimension defined in Fig. 1, Q_d is the constant downstream

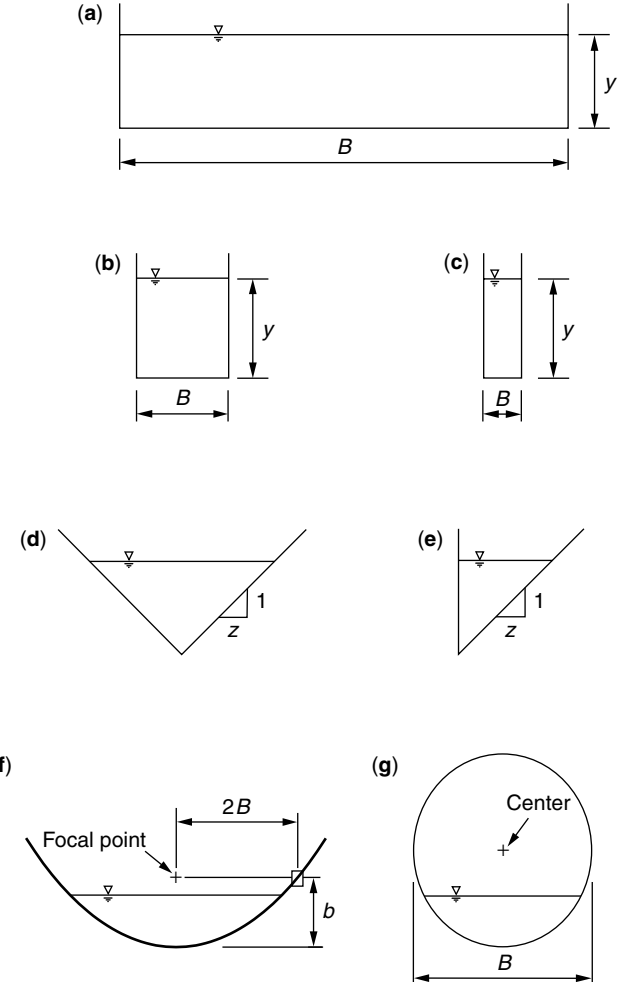


Figure 1. Channels of different cross section: (a) wide rectangular ($B \gg y$); (b) square ($B = y$); (c) deep rectangular ($B \ll y$); (d) triangular; (e) vertical curb; (f) parabolic; (g) circular.

outflow from the channel, and Q_u is the constant upstream inflow to the channel.

Further, Wong (23) showed that the time of travel in a general rectangular channel (i.e., a rectangular channel that has any flow depth) is

$$t_t = 0.0167 \left(\frac{n_c}{\sqrt{S_c}} \right)^{3/4} L_c \left[\frac{(1 + 2\mu)^2}{\mu} \right]^{1/4} \left(\frac{Q_d^{3/4} - Q_u^{3/4}}{Q_d - Q_u} \right) \quad (23)$$

where μ is a parameter relating flow depth y to channel breadth B as follows:

$$\mu = \frac{y}{B} \quad (24)$$

Based on Manning's equation, the value of μ can be determined from

$$\left[\frac{\mu^5}{(1 + 2\mu)^2} \right]^{1/3} = \frac{n_c Q}{B^{8/3} \sqrt{S_c}} \quad (25)$$

where Q is the discharge in the channel. Equation 8 can be applied only to channels where the variation in μ is small. For channels that have a large variation in μ , the channel can be divided into longitudinal segments, and the time of travel is determined for each segment.

The units for Eqs. 16–25 are min for t_t ; m m^{-1} for S_c ; m for L_c , B , and y ; and $\text{m}^3 \text{s}^{-1}$ for Q_d , Q_u , and Q . In the application of Eqs. 16–23, the values of n_c may be selected from those recommended by Chow (24, pp. 110–113) or Acrement and Schneider (25).

DESIGN DISCHARGE FOR A V-SHAPED BASIN

Consider a V-shaped basin comprised of two identical overland planes and a drainage channel, as shown in Fig. 2. The two planes are subject to a uniform intensity i_n , and the runoff from the planes becomes the lateral inflow to the channel. The time of concentration for the basin, t_b , is the summation of the overland time of concentration, t_o , and the time of travel in channel, t_t , as follows (8, p. 90)

$$t_b = t_o + t_t \quad (26)$$

The overland time of concentration, t_o , is given by Eq. 12. Based on Eqs. 16–23, the times of travel in the channel, t_t , are as follows:

1. Wide rectangular channel

$$t_t = 7 \left(\frac{B}{2i_n L_o} \right)^{2/5} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/5} \quad (27)$$

2. Square channel

$$t_t = \frac{1.26}{(2i_n L_o)^{1/4}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/4} \quad (28)$$

3. Deep rectangular channel

$$t_t = \frac{0.0265}{B^{2/3}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right) \quad (29)$$

4. Triangular channel

$$t_t = \frac{1.03}{(2i_n L_o)^{1/4}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/4} \left(\frac{z^2 + 1}{z} \right)^{1/4} \quad (30)$$

5. Vertical curb channel

$$t_t = \frac{0.863}{(2i_n L_o)^{1/4}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/4} \left(\frac{1 + \sqrt{z^2 + 1}}{\sqrt{z}} \right)^{1/2} \quad (31)$$

6. Parabolic channel

$$t_t = \frac{2.84 B^{2/13}}{(2i_n L_o)^{4/13}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{9/13} \quad (32)$$

7. Circular channel

$$t_t = \frac{0.594}{B^{2/15} (2i_n L_o)^{1/5}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{4/5} \quad (33)$$

8. General rectangular channel

$$t_t = \frac{0.726}{(2i_n L_o)^{1/4}} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/4} \left[\frac{(1 + 2\mu)^2}{\mu} \right]^{1/4} \quad (34)$$

The units are min for t_t ; mm h^{-1} for i_n ; m for L_o , L_c , B , and y ; and m m^{-1} for S_c .

For the V-shaped basin, the maximum peak discharge may occur under the condition of full-area or partial-area contribution. The maximum discharge is the design discharge for the basin. For planes subject to uniform net rainfall intensity and for full-area contribution, the design rainfall intensity, i_b , can be derived by equating the time of concentration of the basin, t_b , Eq. 26 to the net rainfall

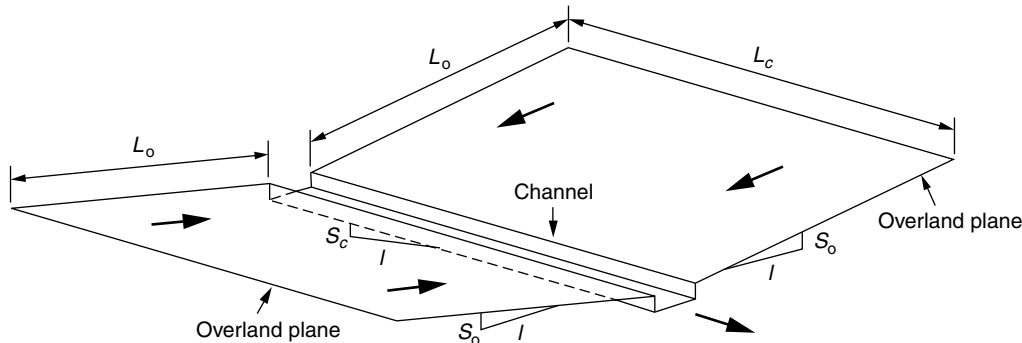


Figure 2. A V-shaped drainage basin.

duration, t_n Eq. 7. The peak discharge for the basin, Q_b , is then derived from i_b as follows:

$$Q_b = \frac{2i_b L_o L_c}{3.6 \times 10^6} \quad (35)$$

The units are $\text{m}^3 \text{s}^{-1}$ for Q_b , mm h^{-1} for i_b , and m for L_o and L_c . Equation 35 can also be used to estimate the peak discharge under the partial-area contribution (i.e., only from the downstream portion of the basin).

Further, for a V-shaped basin that has a wide rectangular channel, an explicit expression for the peak discharge can be derived. The design rainfall intensity, i_b , can be obtained by substituting Eqs. 12 and 27 in Eq. 26 to obtain t_b and equating it to t_n Eq. 7. Substituting i_b in Eq. 35 results in the following explicit expression for the peak discharge Q_b :

$$Q_b = \frac{2L_o L_c}{3.6 \times 10^6} \left\{ \frac{a_b^{1/b_b}}{7 \left[\left(\frac{n_o L_o}{\sqrt{S_o}} \right)^{3/5} + \left(\frac{B}{2L_o} \right)^{2/5} \left(\frac{n_c L_c}{\sqrt{S_c}} \right)^{3/5} \right]} \right\}^{b_b/[1-(2b_b/5)]} \quad (36)$$

where a_b and b_b are the respective values of a_n and b_n in Eq. 7 for $i_n = i_b$ and $t_n = t_b$. The units are $\text{m}^3 \text{s}^{-1}$ for Q_b ; m for L_o , L_c , and B ; and m m^{-1} for S_o and S_c .

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KINEMATIC WAVE AND DIFFUSION WAVE THEORIES

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A wide range of phenomena, natural as well as man-made, in physical, chemical, and biological hydrology exhibit characteristics similar to those of kinematic or diffusion waves. The wide range of phenomena suggests that these

waves are very pervasive. This study presents theories that are considered fundamental to advancing the state of the art of hydrology.

INTRODUCTION

The term “wave” implies a disturbance traveling upstream, downstream, or remaining stationary. We can visualize a water wave propagating where the water itself stays very much where it was before the wave was produced. We witness other waves that travel as well, such as heat waves, pressure waves, and sound waves. There is obviously the motion of matter, but there can also be the motion of form and other properties of matter.

Flow phenomena, according to the nature of particles composing them, can be classified into two categories: (1) flows of discrete noncoherent particles and (2) flows of continuous coherent particles. Examples of flows in the first category include traffic flow; transport of sand and gravel in pipes, flumes, and rivers; evolution of ripples, dunes, and bars on a river bed; settlement of sediment particles in a tank; and movement of microbial organisms. Exemplifying flows in the second category are overland flow, flood movement, baseflow, snowmelt, movement of glaciers, infiltration, evaporation, solute transport, ion exchange, and chromatographic transport. Because of their wave-like behavior, these flow phenomena can be described by the hydrodynamic wave theory or one of its variants, including the kinematic wave theory, diffusion wave theory, gravity wave theory, and linear wave theory.

A survey of geophysical literature reveals that the kinematic and diffusion wave theories have been applied to a wide spectrum of problems, including watershed runoff modeling, flood routing in rivers and channels, movement of soil moisture, macropore flow, subsurface storm flow, erosion and sediment transport, debris flow, solute transport, ion exchange, chromatography, sedimentation, glacial motion, movement of snowmelt water, flow over porous beds in irrigation borders and furrows, river-ice motion, vertical mixing of coarse particles in gravel-bed rivers, to name but a few. Other fields outside the realm of geophysics that apply the kinematic and diffusion wave theories include traffic flow, movement of agricultural grains in a bin, and blood circulation. It can be argued that within certain limitations a wide range of hydrologic phenomena can be approximated quite closely by kinematic and diffusion wave theories. Outside of these limitations, the theories provide only a crude approximation and the full hydrodynamic wave theory or one of its higher order variants may be needed.

Before presenting the theories, one may ask, What is meant by kinematic and diffusive? Although papers and textbooks in hydrology define kinematic and diffusive waves mostly from an approximation of the momentum equation and occasionally from defining flux laws, a comprehensive account of these terms is lacking in literature. Therefore, a short discussion to that end is in order.

WHAT IS KINEMATICS?

The term “kinematic” involves consideration of such macroscopic properties of a flow phenomenon as velocity,

discharge, concentration, and time of travel. It does not involve consideration of forces and mass. Thus, the connotation of “kinematic” is that it is phenomenological and less physical, but its representation may be derived from or related to dynamic considerations involving forces, mass, momentum, and energy.

Although studies based on kinematics existed long before the development of Greek science, Ampere (1) is credited with introducing the term “kinematics” for the study of motion in his famous essay on the classification of sciences. His intent was to create a discipline for the study of motion without regard to the forces involved. Thus, kinematics is defined as the science that deals with the study of motion in the most general way. The motion can be translatory, dilatational, or rotational (angular) (2).

It may be instructive to view kinematics in the context of geometry and dynamics. From the standpoint of dimensions, geometry implies only length, whereas kinematics involves length and time, and dynamics considers length, mass, and time. Thus, kinematics is somewhere between geometry and dynamics, perhaps closer to the latter. Although kinematics is related to geometry, it can also be studied without geometry by means of numerical description of the functions as the Babylonians seem to have done it—a precursor to curve fitting. In kinematics, we are concerned with the position of a point at a certain time and also with the displacement accomplished and the distance traversed in a certain time interval. Other functions of time such as velocity and acceleration can be introduced. Even higher derivatives such as the rate of change of acceleration, called jerk, and even higher order derivatives can be included.

It should be noted that the introduction of mass does not necessarily make some motions strictly dynamic; they may remain strongly and functionally kinematic. For instance, the formulation of equations for conservation of volume in fluid mechanics is very much in the realm of kinematics. When the equations for conservation of mass are formulated, they are still in the same domain; the two treatments are very similar. Even the discussion of momentum flux and kinetic energy flux are fundamentally kinematic. In general, there are terms in the equations of dynamics that are essentially kinematic. The difference between kinematics and dynamics thus requires close scrutiny.

KINEMATIC WAVE THEORY

In a seminal contribution, in 1955, Lighthill and Whitham developed the kinematic wave theory, and they gave a full account of the theory for describing flood movement in long rivers and traffic flow on long crowded roads (3,4). The theory is comprised of three components: (1) the theory of kinematic continuous waves, (2) the theory of shock waves, and (3) the theory of the formation of shock waves from continuous waves. Iwagaki (5) developed an approximate method of the characteristics for routing steady-state flow in open channels of any cross-sectional shape that have nearly uniform lateral flow and proposed that the method would be applicable to the hydraulic analysis of runoff estimates in river basins. Implicit in his method was the

kinematic wave assumption. Thus, Iwagaki (1955) can also be credited with independently conceiving of the kinematic wave concept and developing the method of kinematic wave routing.

Following Lighthill and Whitham (3,4), kinematic waves exist if it can be assumed with sufficient accuracy that there is a functional relationship between flux, concentration, and position. Thus, the kinematic wave theory is mathematically expressed by the law of conservation of mass through the continuity equation and a flux–concentration relation. The coupling of these two equations leads to a first-order partial differential equation that has only one system of wave characteristics. The essence of this is that the wave property is derived from the continuity equation alone. The governing equation (or continuity equation) is only of the first order, so kinematic waves possess only one system of characteristics. This would also imply that kinematic waves travel only in a downstream direction and that the kinematic wave theory cannot accommodate waves that travel in the upstream direction as in backwater flow. In the absence of any lateral inflow and/or outflow, the flux along the characteristics is constant, and the kinematic waves are nondispersive and nondiffusive. However, the waves are dispersive and diffusive when there is lateral inflow and/or outflow, as, for example, in rainfall runoff.

Continuous Waves

If the volume of a quantity passing a given point x in unit time is denoted by flux F and the concentration C is defined as the volume of the quantity per unit distance, then for a one-dimensional flow system without any lateral sources and sinks, the continuity equation or the mass conservation law is

$$\frac{\partial C}{\partial t} + \frac{\partial F}{\partial x} = 0 \quad (1)$$

Equation 1 is first order and states that the volume of the quantity in a small element of length changes at a rate equal to the difference between the rates of inflow and outflow.

Fundamental to the development of the kinematic wave theory is the development of a flux–concentration relation. A general flux–concentration relation can be assumed as

$$F = F(C, x) \quad (2)$$

Equations 1 and 2 have two unknowns, F and C , and can therefore be combined into one equation having one unknown. Multiplying Eq. 1 by

$$c = \left(\frac{\partial F}{\partial C} \right)_{x \text{ constant}} = c(C, x) \quad (3)$$

one obtains

$$\frac{\partial F}{\partial t} + c \frac{\partial F}{\partial x} = 0 \quad (4)$$

Equation 4 is the kinematic wave equation. The quantity c has various names. In hydraulics, it is called wave velocity or celerity, and it is referred to as mobility in vadose zone

hydrology. It is the slope of the flux–concentration relation at a fixed x . This is not the same as the mean velocity u which is expressed as

$$u = F/C \quad (5)$$

The relation between c and u is thus obvious:

$$c = \frac{d}{dC}(uC) = u + C \frac{du}{dC} \quad (6)$$

Equation 6 shows that c is greater than u , if u increases with C as in river flow; it is less than u if u decreases with C as in sedimentation; and it is equal to u if u does not change with C . Equation 4 states that F is constant in waves traveling past a point whose celerity c is given by Eq. 3. The kinematic wave equation 4 has one system of characteristics traveling only in the downstream direction, given by $dx = c dt$, and along each of these characteristics, F is fixed.

Although Eq. 3 constitutes the basic building block of the kinematic wave theory, a more popular, but more restrictive, derivation of the theory in hydrology is obtained by assuming that the local acceleration, convective acceleration, and pressure gradient (or concentration gradient) in the momentum equation are negligible. This assumption states an equivalence between frictional and gravitational forces. When only the local and convective acceleration terms are assumed negligible, the remainder of the momentum equation (i.e., the pressure gradient equaling the difference between gravity and frictional slopes) leads, in conjunction with the continuity equation, to diffusion wave theory. When the full momentum equation is used jointly with the continuity equation, the result is the dynamic wave theory. In general, the local acceleration and the convective acceleration are of the same order of magnitude but are opposite in sign, thus counteracting each other. In a wide range of problems in hydrology, the frictional and gravitational terms are dominant, and this is one of the reasons for the popularity of the kinematic wave theory.

Formation of Kinematic Shocks

A kinematic shock is a discontinuity representing a sudden rise or surge in flow depth. For example, during wave movement, faster moving waves overtake slower moving waves, and at a fixed position, there will be an increase in flux and concentration as functions of time, leading to shock formation. Thus, flood waves have an intrinsic, nonlinear tendency to steepen as they propagate downstream, eventually forming a shock. In the characteristic plane, shock formation results from the intersection of characteristics. After some time, the shock weakens and dissolves into a region of uniform flow. Shocks arise in a variety of hydrologic processes. A short but general discussion of shocks is relevant, however, and is given here. Shocks or discontinuities are of first or second order. A discontinuity of the first kind in the concentration is defined by a sudden change of concentration at a certain level. In such a case, the differential equation of continuity (Eq. 4) no longer applies;

instead, it is replaced by an equation stating that the flow of mass into one side of the shock is equal to that on the other side of the shock. Denoting the velocity of discontinuity by U , mass conservation leads to

$$U = \frac{F_2 - F_1}{C_2 - C_1} \quad (7)$$

where subscripts 1 and 2 denote the quantities ahead and behind the shock. Equation 7 shows that, in general, the discontinuity is not at rest but moves with velocity U which is the slope of the chord joining the points (F_1, C_1) and (F_2, C_2) on the F - C diagram.

A discontinuity of the second kind is represented by a very small change in concentration. If $C_2 - C_1 = dC$ is small, the expression for U becomes $dF/dC = c$, the celerity. In this case the velocity of a discontinuity between concentrations C and $C + dC$ is the same as the celerity c . A small change, if maintained, is propagated through the medium of concentration C at velocity c similarly to the propagation of sound through air at a definite velocity. A line of constant concentration, therefore, describes the motion of a boundary between media of concentrations C and $C + dC$ so that its slope is necessarily equal to c . The complete modification of concentration at a position as well as its profile can be characterized as a series of small discontinuities propagated through the medium.

Determination of Kinematic Shocks

Many factors affect the formation of shocks, and these factors can broadly be distinguished as of three types: (1) initial and boundary conditions, (2) lateral inflow and outflow, and (3) watershed geometric characteristics. A full account of the theory of shock formation was given by Lighthill and Whitham (3,4), and a mathematical treatment of the formation and decay of shocks was given by Lax (6).

Carrier and Pearson (7) developed a method for determining the enveloping curve for any region of intersecting characteristics. Croley and Hunt (8) analytically determined boundary-dependent shocks in planar flow. Hair-sine and Parlange (9) dealt with kinematic shocks on curved surfaces. Kibler and Woolhiser (10) investigated geometry-dependent shocks using a kinematic cascade. They derived the mathematical properties of a kinematic cascade and developed a criterion, based on the properties of adjacent flow-plane pairs, to predict when a shock would occur in a cascade. They also developed a numerical procedure for shock fitting. General properties of shock waves along with continuous kinematic waves were discussed. Full equations of motion were employed to investigate the structure of the kinematic shock.

Building on the work of Kibler and Woolhiser (10), Schmid (11) investigated the effect of lateral inflow and outflow as well as geometry on the development of shocks. He derived a generalized criterion for shock formation in an infiltrating cascade that also encompasses cascades that have time-dependent rates of effective rainfall. Using simulated conditions, Ponce and Windingland (12) determined flow and channel characteristics that either tend to promote or inhibit development of kinematic shocks.

Borah (13) and Borah et al. (14) developed approximate but efficient numerical methods for determining the shock path and shock fitting.

DIFFUSION WAVE THEORY

The diffusion wave theory is comprised of (1) the formation of continuous waves, (2) the formation of shocks, and (3) the propagation of shock waves. The diffusion wave formulation is obtained from the continuity equation and a diffusive flux law. In surface water hydrology, the diffusive flux law is obtained by neglecting the inertial terms in the momentum equation, whereas in groundwater and subsurface water hydrology, it is expressed by Darcy's law. When the two equations are combined, the result is a second-order partial differential equation. The second-order differential term is associated with a coefficient expressed in terms of what is referred to as diffusivity. This term introduces diffusion. The first-order spatial derivative and the coefficient associated with it together lead to steepening or breaking of waves. Clearly, these two terms have opposite tendencies of nonlinear wave steepening and diffusion. The diffusivity term is always positive.

When a shock is formed, it can be discontinuous or continuous. If the diffusivity is small, the shock is a rapid but continuous increase in flow concentration over a narrow range. Breaking due to nonlinearity is balanced by diffusion in the narrow range to yield a steady-state profile. The sign of the derivative of the concentration plays an important role in determining whether the shock will break forward or backward.

FLUX LAWS

The term "flux" is defined in two ways in hydraulics. First, flux denotes a volume of any quantity per unit area per unit time. Thus, its dimensions are L/T if the quantity is, say, runoff. Second, flux is defined as any quantity per unit time. For example, volume flux is the volume of a quantity per unit time as exemplified by discharge, mass flux is mass per unit time, momentum flux (it has the same dimensions as force) is momentum per unit time, energy flux (kinetic energy flux has the same dimensions as power) is energy per unit time, and so on. Both definitions are used in water and environmental engineering.

Flux laws are fundamental to the development of transport theories. The flux laws most common in environmental and water sciences are of two types: (1) power laws and (2) gradient laws. Kinematic wave flux laws are the most popular flux laws of the power type, and Darcy's law and the Darcy-Buckingham law are the most popular gradient type flux laws. Generalized flux laws that specialize into power and gradient laws are also used. An example of a generalized flux law is Burgers' law or a generalized version thereof (15). Power type flux laws lead to kinematic wave equations, whereas the gradient type flux laws lead to diffusion waves governed by elliptical or parabolic partial differential equations.

Algebraic Laws

A general expression for the flux–concentration relation is given by Equation 2. One of its special forms is the popular kinematic wave flux law expressed as

$$F = \alpha C^n \quad (8)$$

where α is a parameter and n is an exponent. The meaning and interpretation of α and n may vary with the problem to which Eq. 8 is applied. Equation 8 leads to the famous Chezy and Manning equations which are popular in studies of overland flow and flood routing in open channels and rivers. At a fixed location x , $(\partial F / \partial C)$ defines the wave celerity $c : c = (\partial F / \partial C)|_{x \text{ fixed}}$ which is not the same as the average velocity of flow. When $n = 1$, Eq. 8 becomes

$$F = \alpha C \quad (9)$$

This is a linear flux–concentration law used to describe the movement of meltwater runoff from snowpack.

Beven (16) employed two somewhat uncommon three-parameter forms of the flux–concentration relation for his channel network routing model:

$$F = \frac{C(a + bk) - k}{1 - bC} \quad (10)$$

and

$$\frac{F}{\alpha(1 - \exp(-kF)) + bF} = C \quad (11)$$

where a , b , and k are parameters.

Another flux law is defined as

$$F = \alpha C^2 (C_o - C) \quad (12)$$

where C_o is the maximum value of C and α is a parameter. Equation 12 is used to describe the settlement of particles in a dispersion. Still another flux law used to describe the movement of sediment in flumes and pipes is

$$F = u_0 C \left(1 - \frac{C}{C_0}\right) \quad (13)$$

where u_0 is the velocity of a single particle when there are no other particles in the flow. All of these flux laws lead to kinematic waves.

Gradient Laws

The gradient-type flux law is expressed as

$$F = G(\partial C / \partial x) \quad (14)$$

where G is some function. As an example,

$$F = \beta (\partial C / \partial x)^m \quad (15)$$

where β is a parameter and m is an exponent. Equation 15 is a nonlinear version of Darcy's law. When $m = 1$, Eq. 15 becomes

$$F = \beta (\partial C / \partial x) \quad (16)$$

Equation 16 is Darcy's law, where β describes the negative of the saturated hydraulic conductivity and C the hydraulic head.

A more general form of Eq. 16 arises when β depends on C :

$$F = \beta(C) \partial C / \partial x \quad (17)$$

Equation 17 is the Darcy–Buckingham law where C describes the hydraulic head and β the negative of the unsaturated hydraulic conductivity which varies with moisture content. When a gradient flux law is coupled with the continuity equation, the resulting partial differential equation turns out to be of parabolic type under transient flow conditions and of elliptical type under steady-state conditions.

Generalized Flux Laws

A generalized flux law is obtained by combining the power and gradient-type flux laws. In general,

$$F = G(C, \partial C / \partial x, x) \quad (18)$$

where G is some function and its argument is defined by C , $\partial C / \partial x$, and x . One form of Eq. 18 is

$$F = \alpha C^n + \beta (\partial C / \partial x)^m \quad (19)$$

where m and n are exponents. Equation 19 specializes into (15)

$$F = \alpha C^n + \beta (\partial C / \partial x) \quad (20)$$

and

$$F = \alpha C + \beta \partial C / \partial x \quad (21)$$

Equation 21 is Burgers' flux law (17) used in turbulence modeling; when used in the continuity equation, the resulting partial differential equation is Burgers' equation. Equations 19 and 20 have been employed to rout flows in open channels (15).

VALIDITY OF KINEMATIC AND DIFFUSION WAVE THEORIES

Lighthill and Whitham (3) have shown that, for Froude numbers below 1 (appropriate to flood waves), dynamic waves are rapidly attenuated and kinematic waves become dominant. Using a dimensionless form of the Saint-Venant (SV) equations, Woolhiser and Liggett (18) obtained the kinematic wave number, K , as a criterion for evaluating the adequacy of the kinematic wave (KW) approximation. For K greater than 20, the kinematic wave approximation was considered an accurate representation of the SV equations in modeling overland flow. Morris and Woolhiser (19) modified this criterion by explicitly including a Froude number corresponding to normal flow, F_o , and showed, based on numerical experimentation, that $F_o^2 K \geq 5$ is a better indicator of the adequacy of the kinematic wave approximation.

Using a linear perturbation analysis, Ponce and Simons (20) derived properties of the kinematic wave,

diffusion wave, and dynamic wave representations in modeling open channel flow. Menendez and Norscini (21) extended the work of Ponce and Simons (20) by including the phase lag between the depth and velocity of flow. Based on the propagative characteristics of a sinusoidal perturbation, they derived criteria to evaluate the adequacy of kinematic wave and dynamic wave approximations. Daluz Vieira (22) compared solutions of the SV equations with those of the kinematic wave and dynamic wave approximations for a range of F_O and K and defined regions of validity of these approximations in the $K - F_O$ space. Fread (23) developed criteria for defining the application range of the kinematic wave and dynamic wave approximations. Ferrick (24) defined a group of dimensionless-scale parameters to establish the spectrum of river waves, using continuous transitions between wave types and subtypes.

It is possible to make inferences about the diffusion wave theory using the Vedernikov number, N , which is the ratio of relative celerity of kinematic waves to the relative celerity of diffusive waves. If N is less than one, flow is stable, and if N is greater than one, it is unstable. Thus, $N = 1$ gives the condition of neutral stability. Clearly, a surface disturbance attenuates in stable flow, amplifies in unstable flow, and undergoes no change in neutral stable flow.

The comparative studies cited before, show that many cases satisfy the conditions for the validity of kinematic wave theory and that in surface water hydrology, kinematic waves dominate in such cases. Other wave types may exist but they are either short-lived or play a minor role. This is further elaborated by considering two special cases: (1) uniform, unsteady-state flow and (2) steady-state, nonuniform flow.

Unsteady-State Uniform Flow

Using simplified conditions, Singh (2,25–30) derived error equations specifying error as a function of time for the kinematic wave approximation of space-independent flows. He considered four different types of scenarios depending upon the presence of lateral inflow or rainfall and infiltration: (1) Lateral inflow is constant, and there is no infiltration; if there is, it is included in lateral inflow. (2) Both lateral inflow and infiltration are considered constant. (3) Both lateral inflow and infiltration are included, but their difference is zero. (4) There is no lateral inflow, but infiltration is included. These scenarios were analyzed under two types of initial conditions: (1) the plane or channel is initially wet, and (2) the plane or channel is initially dry. In all, 18 cases were analyzed.

For space-independent flow, the continuity equation takes the form,

$$\frac{dh}{dt} = i - f \quad (22)$$

and the momentum equation takes the form,

$$\frac{du}{dt} = g(S_0 - S_f) - \frac{iu}{h} \quad (23)$$

where g is the acceleration of gravity, i is the lateral inflow (or rainfall intensity), f is the infiltration rate, S_0 is the

bed slope, and S_f is the slope of the energy line. For a kinematic wave approximation, Eq. 23 becomes

$$S_0 = S_f \quad (24)$$

which can be expressed as

$$u = \left(\frac{S_0}{\beta} \right)^{0.5} h^{0.5} \quad (25)$$

where β is a parameter. Singh (2,26,27,29,30) derived error equations for all cases which turned out to be Riccati equations of the form,

$$\frac{dE}{d\tau} = C_0(\tau) + C_1(\gamma, \tau)E + C_2(\gamma, \tau)E^2 \quad (26)$$

where E is the error defined as (kinematic wave solution-dynamic wave solution)/dynamic wave solution, τ is dimensionless time, γ is a dimensionless parameter, and $C_i, i = 1, 2$, and 3, are Riccati coefficients. The parameter γ is analogous to a kinematic wave parameter and is defined as

$$\gamma = \frac{4g^2\beta S_0 h_0}{i_0^2} \quad (27)$$

or

$$\gamma = 4\beta g S_0 \quad (28)$$

depending upon whether or not lateral inflow (i_0) is included. In general, the kinematic wave approximation is very good if γ is equal to or greater than 10. As time progresses, that is, $\tau \geq 10$, the kinematic wave approximation converges to the dynamic wave solution.

Steady-State Nonuniform Flow

Steady-state nonuniform flows are encountered in a variety of natural situations. In overland flow, the steady state is attained for constant rainfall after the flow depth at the outlet has reached equilibrium. This same is true for channel flow subject to constant lateral inflow. For a channel receiving a constant inflow of long duration at its upstream boundary, the flow at the downstream end would reach equilibrium. The steady-state solution aids in understanding the nature of the surface water profile. It may help define the condition for use of zero depth in place of zero influx at the upstream boundary. When the rainfall duration is much longer than the time of equilibrium, steady-state surface water profiles are very useful. Pearson (31) examined the criteria for using the kinematic wave approximation of the SV equations for shallow water flow. For steady-state one-dimensional flow over a plane, he derived a new criterion as $K \geq 3+5/F_O^2$, where K is the kinematic wave number and F_O is the Froude number corresponding to normal flow.

Govindaraju et al. (32,33) provided a comprehensive discussion of the DW theory for steady-state nonuniform flow. They presented both numerical and analytical results for flux-type downstream boundary conditions and using zero inflow at the upstream boundary. Parlange et al. (34) investigated errors in the KW and diffusion wave (DW) approximations by comparing their predictions with the

numerical solution of the SV equations under steady-state conditions. They suggested splitting the solution into two regions, one near the downstream end of the plane and the other covering most of the plane. Singh and Aravamuthan (35–38) derived errors in the kinematic and diffusion wave approximations under four conditions: (1) zero flow at the upstream boundary, (2) finite depth at the upstream boundary, (3) critical flow depth at the downstream end, and (4) zero-depth gradient at the downstream boundary. For economy of space, only the KW case will be discussed. Depending on the inclusion of lateral inflow and infiltration, 21 cases were analyzed for the four different scenarios mentioned before. By comparing the kinematic wave solution with the dynamic wave solution, error equations were derived for all cases. For time-independent flow, the governing equations are

$$\frac{dh}{dx} = i - f \quad (29)$$

and

$$\frac{d}{dt} \left(\frac{1}{2} u^2 + gh \right) = g(S_o - S_f) - \frac{iu}{h} \quad (30)$$

The error equations were generalized Riccati type equations:

$$\frac{dE}{dx} = C_0 E + C_1 E^2 + C_2 E^3 + C_3 E^4 + C_4 E^5 \quad (31)$$

where the parameters, C_i , $i = 1, 2, 3$, and 4, are nonlinear functions of x , h , K , and F_O which are as defined before. In most cases, it was found that the downstream boundary exercised significant influence on the adequacy of the kinematic wave approximation. Away from the boundaries, that is, $0.1 \leq x/L \leq 0.9$, where L is the length of the plane, the kinematic wave approximation was a good approximation (39).

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KINEMATIC WAVE FLOW ROUTING

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Flow routing on planes, in channels, and in storm sewers using the kinematic wave theory is quite popular, especially for urban drainage design and planning. The small watershed sizes and the often fairly steeper prismatic conduits in an urban environment are well suited for application of the theory. Little or no stream-flow data are required to develop the kinematic wave routing method. In nonurban environments, the kinematic routing wave method, corrected for dynamic effects, is also becoming increasingly popular.

KINEMATIC WAVE FORMULATION FOR CHANNEL FLOW ROUTING

The kinematic wave equation is derived from the continuity equation,

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} = 0 \quad (1)$$

and can be expressed in terms of the discharge, Q , as the dependent variable for a channel:

$$\frac{1}{c} \frac{\partial Q}{\partial t} + \frac{\partial Q}{\partial x} = 0, \quad 0 \leq x \leq L \quad (2)$$

where A is the area of flow cross section, c is the travel speed of the flood wave called the kinematic wave speed and may, at a particular cross section and for a given Q , be evaluated from the Kleitz–Seddon law,

$$c = \frac{dx}{dt} = \left(\frac{dQ}{dA} \right) \Big|_{x \text{ fixed}} = \frac{1}{B} \left(\frac{dQ}{dh} \right) \Big|_{x \text{ fixed}} \quad (3)$$

where B is the channel flow width, h is the depth of flow, and A is the flow cross-sectional area. When c is constant, Eq. 2 becomes linear and will have a translatory solution. For a variable c , it is a nonlinear equation. In practice, c is usually estimated from a rating curve relating discharge to depth of flow, expressed in general by the Chezy or Manning formula:

$$Q = CAR^m S_f^{0.5} \quad (4)$$

where R is the hydraulic radius, C is an empirical resistance coefficient, S_f is the frictional slope = bed slope, S_o , and m is an exponent. Because R depends on A , Eq. 4 is usually written as

$$Q = \alpha A^n, \quad \alpha = C(S_f)^{0.5}, \quad n \in [1, 3] \quad (5)$$

where α is the resistance parameter and n is an exponent. Therefore,

$$c = n\alpha A^{n-1} = n \frac{Q}{A} = nu \left(\frac{Q}{\alpha} \right)^{(n-1)/n} \quad (6)$$

where u is the mean flow velocity.

To solve Eq. 2, the initial and boundary conditions are usually assumed as

$$Q(x, 0) = Q_0(x) \quad (7)$$

$$Q(0, t) = Q_u(x), \quad Q_u(0) = Q_0(0) \quad (8)$$

If $Q_0(x) = 0$, $0 < x < L$, then the channel is initially dry. $Q_u(t)$ is such that an analytical solution is not tractable except when $Q_u(t) = Q_u = \text{constant}$. Therefore, numerical solutions with attendant discretization of the solution domains are employed. A number of numerical schemes have been proposed to solve the kinematic wave equations (1). Figure 1 shows a finite-difference grid for a computational cell in (x, t) space. The discretization, however, has its own flaws. It is known that numerical solutions of Eq. 2 introduce variable amounts of numerical diffusion, and the solution resembles a diffusion wave rather than a kinematic wave (2). In first-order numerical schemes, the numerical diffusion is uncontrolled, making the solution depend on the grid size. In second-order schemes, the numerical diffusion vanishes, but a certain amount of numerical dispersion (of the third order) exists (3). Some workers (4) argue that a small amount of numerical diffusion should be welcome, for the kinematic wave solution

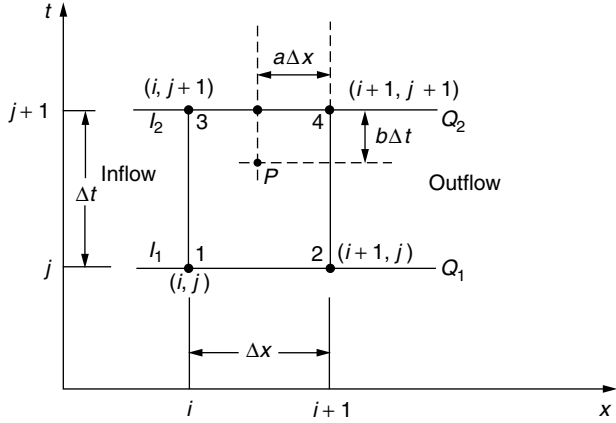


Figure 1. Finite-difference grid for a computational cell in the (x, t) space; a and b are weighting factors in space and time, respectively.

stands to benefit from it and is then applicable to a wider class of problems. Therefore, complete elimination of diffusion may not be desirable from a practical standpoint, for natural diffusion is inherent in physically realistic flows. Thus, one way to accomplish the twin objective of achieving a kinematic wave solution and an acceptable amount of diffusion is to match physical and numerical diffusion judiciously. This procedure has been successfully used for stream-flow routing (5–7). Three possible fully off-centered schemes, which are frequently used, are shown in Fig. 2. The stability depends on the scheme used.

GENERALIZED METHOD FOR A NUMERICAL SOLUTION

A fairly general and flexible finite-difference formulation of the kinematic wave model given by Eq. 2 can be developed, and it can be shown that several existing models are special cases of this formulation. To that end, we employ a rectangular $x-t$ grid, as shown in Fig. 1,

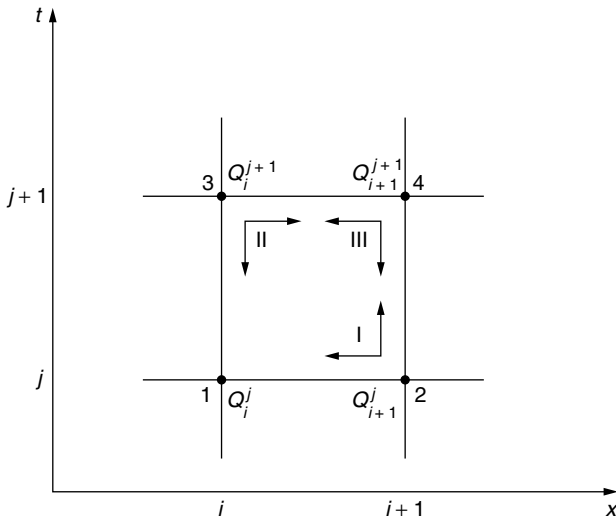


Figure 2. Three fully off-centered numerical schemes for solving kinematic wave equations.

where a ($0 \leq a \leq 1$) and b ($0 \leq b \leq 1$) are the weighting factors in space and time directions, respectively. Let p be a point in the $x-t$ grid around which the finite-difference approximation is attained. If $W(x, t)$ is any variable denoting either $A(x, t)$ or $Q(x, t)$, then,

$$\frac{\partial W}{\partial x} = \frac{\Delta W}{\Delta x} = \frac{1}{\Delta x} [(1-b)(W_{i+1}^j - W_i^j) + b(W_{i+1}^{j+1} - W_i^{j+1})] \quad (9)$$

and

$$\frac{\partial W}{\partial t} = \frac{\Delta W}{\Delta t} = \frac{1}{\Delta t} [(1-a)(W_{i+1}^{j+1} - W_{i+1}^j) + b(W_i^{j+1} - W_i^j)] \quad (10)$$

where subscript i denotes the space coordinate x and j denotes the time coordinate t .

Similar to Eqs. 9 and 10, Eq. 1 can be written in finite difference around a point p as,

$$\begin{aligned} & \frac{1}{\Delta t} [(1-a)(A_{i+1}^{j+1} - A_{i+1}^j) + a(A_i^{j+1} - A_i^j)] \\ & + \frac{1}{\Delta x} [(1-b)(Q_{i+1}^j - Q_i^j) + b(Q_{i+1}^{j+1} - Q_i^{j+1})] = 0 \end{aligned} \quad (11)$$

Depending on the values of a and b , point p may lie anywhere within or on the boundaries of the rectangular grid element. Usually Q_{i+1}^{j+1} and A_{i+1}^{j+1} are unknowns, and these at previous points (i, j) , $(i+1, j)$, and $(i, j+1)$ are known. Arranging the unknowns on the left side and knowns on the right side, Eq. 11 can be written as

$$\begin{aligned} bQ_{i+1}^{j+1} + (1-a)A_{i+1}^{j+1} \frac{\Delta x}{\Delta t} &= bQ_{i+1}^j + (1-a)A_{i+1}^j \frac{\Delta x}{\Delta t} \\ &- Q_{i+1}^j - bQ_i^j + aA_i^j \frac{\Delta x}{\Delta t} + bQ_i^{j+1} - a \frac{\Delta x}{\Delta t} A_i^{j+1} \end{aligned} \quad (12)$$

The kinematic wave assumption requires a unique relation between Q and A as in Eq. 5. Hence, there remains actually only one unknown on the left side of Eq. 12. Let

$$f(Q_i^j) = bQ_i^j - aA_i^j \frac{\Delta x}{\Delta t} \quad (13)$$

and

$$g(Q_i^j) = bQ_i^j + (1-a)A_i^j \frac{\Delta x}{\Delta t} \quad (14)$$

Taking advantage of Eqs. 13 and 14, Eq. 12 can be expressed as

$$g(Q_{i+1}^{j+1}) = g(Q_{i+1}^j) + f(Q_i^{j+1}) - f(Q_i^j) + Q_i^j - Q_{i+1}^j \quad (15)$$

Using the knowledge of the Q - A relationship, functions $f(Q)$ and $g(Q)$ can be computed. First, the right side of Eq. 15 is evaluated. Then, Q_{i+1}^{j+1} is evaluated by using the Q - A relationship. Thus, calculations can proceed either in the x direction or the t direction. The generalized scheme is stable for $a/b \leq c_n \leq (1-a)/(1-b)$, where c_n is the Courant number $= c\Delta t/\Delta x$. When $a = b = 0.5$, the scheme is second-order accurate. This means that point p is centrally located in the rectangular grid. As p departs from the center or as a and b depart from the value of 0.5, truncation errors of the order $O(\Delta x)$ and $O(\Delta t)$ increase,

respectively and independently. This truncation is the cause of the numerical diffusion encountered in numerical kinematic channel flow routing. Thus, the desired level of numerical diffusion can be achieved by an appropriate choice of a and b values. A more complete account of the stability and convergence properties of the generalized scheme is given by Smith (8), who showed that several flood routing models are special cases of Eq. 15.

Reservoir Flow Routing

The routing of flow through a reservoir involves a volume balance and the assumption that reservoir storage is controlled by outflow. The volume balance can be expressed as

$$\text{Rate of change of storage} = \text{inflow} - \text{outflow},$$

$$(A_{i+1}^{j+1} - A_{i+1}^j) \frac{\Delta x}{\Delta t} = \frac{1}{2}(Q_i^j + Q_{i+1}^{j+1}) - \frac{1}{2}(Q_{i+1}^j - Q_{i+1}^{j+1}) \quad (16)$$

By rearranging Equation 16 and comparing Eq. 12, one gets

$$0.5Q_{i+1}^{j+1} + A_{i+1}^{j+1} \frac{\Delta x}{\Delta t} = 0.5Q_{i+1}^j + A_{i+1}^j \frac{\Delta x}{\Delta t} - Q_{i+1}^j - 0.5Q_i^j + Q_i^j + 0.5Q_i^{j+1} \quad (17)$$

Equation 17 is a special case of Eq. 12 where $a = 0$ and $b = 0.5$.

Muskingum Method

Flow routing in channels, using the Muskingum method, is based on Eq. 16 and a relation between storage (S), inflow (I), and outflow (Q) (9):

$$S = K[x_m I + (1 - x_m)Q] \quad (18)$$

where x_m ($0 \leq x_m \leq 1$) is a weighting factor and K is travel time. The right side of Eq. 18 defines prism storage and wedge storage; the latter depends on the amount by which inflow exceeds outflow. Using Eq. 3,

$$K = \frac{\Delta x}{c} = \Delta x \left(\frac{dA}{dQ} \right) = \frac{\Delta x \Delta A}{\Delta Q} \quad (19)$$

In the Muskingum method, c is considered constant for a channel reach. Then,

$$KQ = \Delta x Q \frac{\Delta A}{\Delta Q} = \Delta x \frac{Q}{u} = \Delta x \frac{Q}{c} = \Delta x A \quad (20)$$

Equations 18–20 can be combined with the volume balance equation and then written in finite-difference form, similar to Eq. 12, as

$$0.5Q_{i+1}^{j+1} + (1 - x_m)A_{i+1}^{j+1} \frac{\Delta x}{\Delta t} = 0.5Q_{i+1}^j + (1 - x_m)A_{i+1}^j \frac{\Delta x}{\Delta t} - Q_{i+1}^j - 0.5Q_i^j + x_m A_i^j \frac{\Delta x}{\Delta t} + Q_i^j + 0.5Q_i^{j+1} - x_m A_i^{j+1} \frac{\Delta x}{\Delta t} \quad (21)$$

If $a = x_m$ and $b = 0.5$, Eq. 21 is a special case of Eq. 12. A nonlinear Muskingum method is discussed by Koussis and Osborne (10) and Singh and Scarlatos (11).

Brakensiek's Models

Brakensiek (12) employed the kinematic wave method for flood routing in prismatic channels. He used three numerical schemes which are special cases of Eq. 12 for

$$\begin{aligned} a = 0.5 \text{ and } b = 0.5 & \text{ Model I (centered)} \\ a = 0.5 \text{ and } b = 1.0 & \text{ Model II (implicit)} \\ a = 0.5 \text{ and } b = 0.0 & \text{ Model III (explicit)} \end{aligned}$$

His flood routing results were sensitive to model type. The first two schemes were more stable than the explicit scheme. The use of an implicit scheme did not require simultaneous solution of Q and A at a new point.

HYMO Model

Williams (13) developed a hydrologic modeling method, called HYMO, using a variable travel time routing procedure. The routing employs the volume balance using arithmetic averages to define discharges over a time increment and end areas over a distance increment. This amounts to using $a = b = 0.5$. A distinguishing feature of HYMO is said to be the variable travel time as a function of the unknown discharge, and therefore it involves an iterative calculation. Smith (8), however, has shown that the travel time has to be a constant. The travel time, T , is calculated in HYMO as

$$T = \frac{\text{reach length}}{\text{average velocity}} = \frac{2S}{I + O} \quad (22)$$

where S is the storage based on end areas, I is the inflow, and Q is the outflow. The wave velocity c , therefore, is

$$c = \frac{L}{T} = \frac{1}{2} \frac{I + O}{A_{av}} = \frac{Q_i^j + Q_{i+1}^j}{A_i^j + A_{i+1}^j} \quad (23)$$

This implies that

$$c = \frac{dQ}{dA} = \frac{Q}{A} \quad (24)$$

This means that the wave velocity in HYMO is defined by the $Q - A$ chord rather than by the tangent. This is true only for a linear channel where

$$u(x) = \frac{Q}{A} = f(x) \quad (25)$$

This implies that the velocity, u , will be independent of discharge or storage and travel time must necessarily be a constant.

SSARR Model

The U.S. Army Corps of Engineers (14) uses a continuous-time, stream-flow simulation and reservoir regulation model, called SSARR. The model represents a channel by a number of reaches through each of which storage is essentially routed. This amounts to using Eq. 12 where $a =$

0.0 and $b = 0.5$ for each reach. Flood wave modification, therefore, depends on the number of reaches employed.

SWMM Model

The storm water management model (SWMM), developed by the U.S. Environmental Protection Agency (15), is widely used in the United States for urban storm water analysis. In the channel routing phase, a finite-difference form of the continuity equation is employed using weighting coefficients, w_x and w_t , respectively, in space and time, which essentially are $w_x = 1 - a$, and $w_t = b$. When convective acceleration terms are included in the model, then $a = 0.45$ and $b = 0.55$ are used.

MUSKINGUM-CUNGE METHOD

Cunge (2) proposed an explicit finite-difference scheme for solving Eq. 2, which may also be a basis for a generalized treatment of kinematic wave models. The scheme centers the time derivative by taking $b = 0.5$ and retains the weighting coefficient a in space. The celerity, c , is taken as an average constant value for the reach or computational cell. Equation 2 can be written in finite-difference form as

$$\frac{1}{c\Delta t}[(1-a)(Q_{i+1}^{j+1} - Q_{i+1}^j) + a(Q_i^{j+1} - Q_i^j)] + \frac{1}{2\Delta x}(Q_{i+1}^j - Q_i^j + Q_{i+1}^{j+1} - Q_i^j) = 0 \quad (26)$$

By solving for Q_{i+1} , Eq. 26 can be written as the classical Muskingum equation:

$$Q_{i+1}^{j+1} = C_1 Q_i^j + C_2 Q_i^{j+1} + C_3 Q_{i+1}^j \quad (27)$$

or

$$Q_4 = C_1 Q_1 + C_2 Q_2 + C_3 Q_3$$

where

$$C_1 = \frac{\Delta t + 2aK}{2K(1-a) + \Delta t} \quad (28)$$

$$C_2 = \frac{2K(1-a) - \Delta t}{2K(1-a) + \Delta t} \quad (29)$$

$$C_3 = \frac{\Delta t - 2aK}{2K(1-a) + \Delta t} \quad (30)$$

and

$$K = \frac{\Delta x}{c} \quad (31)$$

$$C_1 + C_2 + C_3 = 1 \quad (32)$$

The unit of K is time, and it has the connotation of storage-delay time, travel time, translation time, or lag time. It can be estimated from either the Kleitz-Seddon law, hydrographic analysis, or observations.

Cunge (2) showed that Eqs. 26–32 constitute a second-order approximation of the diffusion wave equation if the weighting coefficient a is evaluated as

$$a = \frac{1}{2} \left(1 - \frac{Q}{BS_o\Delta x} \right) \quad (33)$$

where S_o is the channel bed slope. Cunge derived Eq. 23a from a Taylor series expression of $Q(x + \Delta x, t + \Delta t)$ in the finite-difference form of Eq. 2 and a comparison with the coefficients of the diffusion wave equation for regular channels.

Equation 33 specializes to the equation of Kalinin and Milyukov (16) for $a = 0$:

$$\Delta x = \frac{Q}{S_o Bc} = \frac{Q}{S_o(dQ/dh)} \quad (34)$$

where h is the flow depth. Equation 34 can be used to compute the length of subreaches for flow routing in a channel reach.

In the traditional Muskingum method, a single value of a or Δx is used for the entire duration of routing. That can be accomplished by evaluating Eq. 33 or 34 for a representative value of Q and corresponding values of B and c , or dQ/dh . In irregular channels, the average values of these parameters for the subreach should be used. The value of a , it is found, depends on Δx .

Ponce (17) proposed a simplified Muskingum routing equation by taking $\Delta t/K$ and a in Eqs. 26–31 as

$$\frac{\Delta t}{K} = 1, a = 0 \quad (35)$$

Then, it follows that

$$C_1 + C_2 + C_3 = \frac{1}{3} \quad (36)$$

leading to the simplified Muskingum routing equation,

$$Q_{i+1}^{j+1} = \frac{1}{3}[Q_i^j + Q_i^{j+1} + Q_{i+1}^j] \quad (37)$$

Physically, Eqs. 35 and 36 fix Δt and Δx , and K is the time taken by a flood wave traveling a distance Δx at celerity c given by Eq. 31. Therefore, the Courant number c_N is

$$c_N = \frac{c\Delta t}{\Delta x} = 1 \quad (38)$$

implying equality between flood-wave celerity c and grid celerity $\Delta x/\Delta t$.

Ponce (17) defined a grid Reynolds number R_g (the ratio of physical or hydraulic diffusivity to numerical diffusivity) from Eqs. 33 and 36 or Eq. 34 as

$$R_g = \frac{Q}{BS_o c \Delta x} = 1 \quad (39)$$

which states that the channel diffusivity $Q/(2BS_o)$ is equal to the grid diffusivity $c\Delta x/2$. In Ponce's method,

Δx and Δt can be computed from flow variables by using Eqs. 35 and 39. For example, Δx is given by Eq. 34 or

$$\Delta x = \frac{q}{cS_o} \quad (40)$$

and Δt by Eq. 39,

$$\Delta t = \frac{\Delta x}{c} = \frac{q}{S_o c^2} \quad (41)$$

where q is the unit-width discharge. For natural (nonprismatic) channels, Eq. 37 relates the channel friction to cross-sectional shape and expresses the steady-state discharge relation. Using Eq. 38 in Eq. 40 yields

$$\Delta x = \frac{A}{nBS_o} \quad (42)$$

and

$$\Delta t = \frac{\Delta x}{c} = \frac{A^{2-n}}{\alpha n^2 BS_o} \quad (43)$$

The routing coefficients C_1 , C_2 , and C_3 can be expressed in terms of c_N and R_g by using Eqs. 31, 33, 38, and 39:

$$C_0 = 1 + c_N + R_g \quad (44)$$

$$C_1 = \frac{1 + c_N - R_g}{C_o} \quad (45)$$

$$C_2 = \frac{-1 + c_N + R_g}{C_o} \quad (46)$$

$$C_3 = \frac{1 - c_N + R_g}{C_o} \quad (47)$$

where c_N and R_g are expressed in terms of B , c , Δx , Δt , and Q .

If there is lateral inflow q_L , the Muskingum equation 27 can be modified as

$$Q_{i+1}^{j+1} = C_1 Q_i^j + C_2 Q_i^{j+1} + C_3 Q_{i+1}^j + Q_L \quad (48)$$

in which

$$Q_L = \frac{2cq_L \Delta t}{(\Delta t/K) + 2(1-a)} \quad (49)$$

where q_L is the unit-width lateral inflow. Because $c = \Delta t/K$ and if $K = \Delta t$ and $a = 0$, Eq. 50 reduces to

$$Q_L = \frac{2q_L \Delta x}{3} \quad (50)$$

It is pertinent at this point to comment on the role of the bed slope S_o and the frictional slope S_f in the Muskingum–Cunge method. In prismatic channels, S_o is used as one of the main channel parameters. Under steady-state, uniform conditions, it is known that $S_o = S_f = S_e$, where S_e is the energy slope. Under steady-state but nonuniform conditions, $S_o \neq S_e = S_f$. In the Muskingum–Cunge method, linearization (or the Taylor series approximation) of the flow is performed around the steady-state uniform or equilibrium flow condition represented by Q_o which permits using S_o in

Eq. 33 or 34. In actual practice, however, S_o is not often readily determined, owing largely to local channel bed irregularities. For such channels, S_f is more readily obtained from a given value of Manning's roughness coefficient. Thus, it is more convenient to use S_f as an estimator of S_o . Ponce (18) used S_f in lieu of S_o in his work on linearized diffusion wave modeling.

MUSKINGUM–CUNGE METHOD WITH VARIABLE PARAMETERS

Ponce and Yevjevich (6) considered the space–time variation of K and a as the flow varied. In this case, Δt is usually fixed, and Δx and S_o are specified for each computational cell comprising four grid points. This requires determining flood wave celerity c and the unit-width discharge q for each computational cell. The Muskingum–Cunge method requires calculating the average values of Q and c for each computational cell to compute K and a . Ponce and Yevjevich (6) state three ways of determining c and q for use in computing C_N and R_g : (1) using a two-point average of values at grid point (i, j) and $(i + 1, j)$; (2) using a three-point average of values at grid points (i, j) , $(i + 1, j)$, and $(i, j + 1)$; and (3) using a four-point average calculation by iteration. The average values for a four-point scheme can be expressed by taking the average at four points (19). Holden and Stephenson (19) showed through numerical experiments that parameter a is virtually independent of grid spacing. Furthermore, the lower the value of a , the greater the attenuation in flood peak. This means that the closer a is to zero, the greater attenuation occurs as numerical diffusion. Although the values of a found are close to 0.5 for many flow conditions, the value of a drops significantly lower for river channels because more gentle bed slopes lead to higher attenuation of a flood hydrograph. The implicit formation of the Muskingum–Cunge method has no particular advantage over the explicit one, and hence the latter may be preferable.

KOUSSIS MODEL

Koussis (20) proposed a generalized numerical solution of Eq. 2 in which discretization was carried out only in space and continuous functions for time derivatives were retained. Considering c as an average value for the computational cell, Eq. 2 was written as an ordinary differential equation in Q_{i+1} . Subject to the initial condition $Q_{i+1}(0) = Q_i(0) = Q(x, t = 0)$, the resulting equation was solved by assuming a linear variation of $Q_j(t)$ over Δt and then expressing it in the form of the Muskingum equation.

MODEL PARAMETERS

Four parameters are involved in the previous kinematic wave models: Δx , Δt , a , and c . These parameters are used to compute the routing coefficients. The parameters can be constant (linear case) or variable (nonlinear case).

Spatial Grid Spacing

A single-valued storage function cannot be prescribed for long river reaches, and floods should be routed by

dividing the reach length L into N subreaches of length Δx . Laurenson (21,22) has shown that Δx and the weighting parameter a are not independent of each other in that a decreases with Δx or as storage becomes less distributed. Because a cannot be negative, Δx must have a lower limit, or N must have a finite upper limit. Nash (23) argued that unrealistic negative outflows at the start of the outflow hydrograph were the result of applying the Muskingum method to distributed storages ($a \neq 0$). Weinmann and Laurenson (1) showed that these negative outflows can be ignored under certain conditions.

Temporal Grid Spacing

If Δt is much smaller than $\Delta x/c$, then the disturbance might not have traveled distance Δx . For $a \neq 0$, computation of variables at the current section may lead to unrealistic results such as a dip in the outflow hydrograph or negative outflows for an initially dry channel. The best time step for the kinematic wave model can be taken as $\Delta t = \Delta x/c$.

Weighting Factor, a

Strictly speaking, a can take on any value from zero to one. The value $a > 0.5$ results in amplifying flood waves and is therefore not important in flood routing. The value $a = 0.5$ for all values of $\Delta x/\Delta t$ results in pure translation of a flood wave (2), corresponding to fully distributed storage. The values $a < 0.5$ introduce numerical diffusion resulting in wave attenuation. For the extreme case, $a = 0$ corresponds to fully concentrated storage or a reservoir that has an instantaneous response and attenuating inflow peak. This discussion then shows that when Δx and Δt are chosen within prescribed limits, a flood wave can be both attenuated and translated by an appropriate choice of a . This is similar to what is done by using lag and route models that treat translation and attenuation separately. Translation can also be achieved by choosing $a = 0$ and an appropriate value of N , when concentrated storages are arranged in series, as in the Kalinin–Milyukov model. Frequently, a constant value of a will be sufficient, unless it varies greatly.

Wave Celerity, c

In kinematic wave models, c for a given reach is a function only of flow depth, for the rating curve is a single-valued function. In the coefficient representation of these models, a flood wave is attenuated by taking $a < 0.5$. Similar to kinematic wave models, the storage routing models commonly employed in hydrology use a single-valued storage–discharge curve. Koussis (5,20,24,25), Williams (13), and Ponce (4), amongst others, included a looped rating curve in kinematic wave models and thereby corrected them for dynamic effects.

To determine the average reciprocal wave celerity, the channel reaches should be selected as morphologically similar. For natural rivers, the variations in channel cross sections, bed slope, and roughness may be so great that it may not suffice to use only two $A(Q)$ curves at i and $(i + 1)$ cross sections, unless Δx is quite small. Nevertheless, it is reasonable to compute the average reciprocal wave celerity

from steady flow $A(Q)$ curves at several cross sections. In prismatic channels, this may be computed from the slope of the $A(Q)$ curve for unsteady-state flow at i and $(i + 1)$ cross sections.

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RELIABILITY CONCEPTS IN RESERVOIR DESIGN

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INTRODUCTION

The reliability of a system is defined as the probability that the system will perform the required function for a specified period of time under stated conditions. Reliability is the complement of the probability of failure (or risk), the probability that the “loading” will exceed the “capacity.” Denoting α as the reliability, β the probability of failure, and $P[\omega]$ the probability of an event ω , the mathematical expression of this definition is

$$\alpha := P[L(t) < C(t); t \in \Pi] =: 1 - \beta \quad (1)$$

where $L(t)$ and $C(t)$ represent the loading and capacity, respectively, at time t , within a certain time period Π (e.g., a year). The failure of a system can be classified as *structural failure* or *performance failure*. Structural failure involves damage of the structure or facility, hindering its ability to function as desired in the future, whereas performance failure does not necessarily involve structural damage but rather inability of the system to perform as desired at some time within the period of interest, which results in temporary unfavorable consequences.

RESERVOIR DYNAMICS

A reservoir's function is to regulate natural inflows, which vary irregularly, to provide outflows at a more regular rate that is determined by water demand for one or more uses (water supply, irrigation, hydropower), temporarily storing the surplus, when inflows exceed outflows. Reservoir dynamics are more conveniently expressed in discrete rather than continuous time.

The quantities that are necessary to describe dynamics are the following:

Storage S_t . More precisely known as *active storage*, it is the volume of water stored, at time t , above the minimum level, which is determined either technically (i.e., as the level of the lowest valve of off-take) or legally by a decree imposing rules for a reservoir's operation. Active storage

S_t ranges between zero and a maximum value c imposed by the reservoir size, which corresponds to the level of the spillway crest (or some specified level above it when the sluice gates are constructed over the spillway). During floods, excess water is routed through the spillway, which causes temporary storage above the normal limit c . This is known as flood control storage. Water storage below the minimum level, known as *dead* or *inactive storage*, serves two main purposes: It provides volume for sediment accumulation and environmental protection, as it protects the habitat of the reservoir during dry periods by hindering complete emptying. Associated with the last function is also conservation of the quality of the landscape. This article is focused on the design of the active storage of a reservoir; some notes on additional storage zones are contained in the last section of the article.

Net Inflow X_t . It is the algebraic sum of cumulative inflows to the reservoir from time $t - 1$ to time t , minus the losses during the same time period. Inflows include runoff from the catchment upstream of the reservoir (typically, the main component of inflows), rainfall on the surface area of the reservoir, and, possibly, water artificially conveyed from other sources (e.g., interbasin transfers through tunnels or pipelines). Losses include evaporation from the surface area of the reservoir, possibly seepage to groundwater, and leakage under or through the dam.

Water Demand, δ_t . It is the sum of all water requirements for the different water uses served by the reservoir for the time period $(t - 1, t)$. The demand may vary with time (e.g., due to seasonal agricultural demand or due to some rule, usually based on the quantity of water in the reservoir).

Release, R_t . Also known as draft, withdrawal, or outflow, it is the actual amount of water taken from the reservoir to satisfy water demand during the time period $(t - 1, t)$. When there is a sufficient amount of water in the reservoir, R_t equals demand δ_t ; otherwise $R_t < \delta_t$.

Spill, W_t . It is the excess water that, during times of floods and simultaneously high reservoir storage, cannot be stored in the reservoir due to the upper reservoir storage limit c .

Reservoir dynamics are easily expressed by the law of mass conservation, or equivalently, the water balance equation. Considering that S_t is limited between 0 and c , the water balance equation is easily formulated as

$$S_t = \max(0, \min(S_{t-1} + X_t - \delta_t, c)) \quad (2)$$

In addition, the release is determined as

$$R_t = \min(S_{t-1} + X_t, \delta_t) \quad (3)$$

and the spill as

$$W_t = S_{t-1} - S_t + X_t - R_t = \max(0, S_{t-1} + X_t - \delta_t - c) \quad (4)$$

Equations 2–4 apply when the inflow and withdrawal occur at constant rates throughout the period $(t, t - 1)$ —this could be called the “steady” model. A simple

modification of the equations allows for the case where the inflow (or withdrawal) is highly seasonal and can (in the limit) be modeled as a sudden occurrence; this could be called the “sudden” model. These two models, called “simultaneous” and “staggered” by Pegram (1), bound all the behavior observed in real reservoirs.

DEFINITION OF RELIABILITY APPLIED TO A RESERVOIR

Now, the previously stated general definition of reliability [also known as dependability; e.g., (2) p. 312] can be applied to a reservoir. It is observed that the failure of a reservoir's function is a performance failure, a failure to meet the water demand. At time t , the loading is the water demand δ_t , and the capacity is the sum of S_{t-1} (storage at time $t - 1$) and X_t (inflow from time $t - 1$ to t). Thus, application of Equation 1 yields

$$\alpha = 1 - \beta = P[\delta_t < S_{t-1} + X_t] \quad (5)$$

Considering Equations 2 and 3, the following equivalent and more convenient expressions are found:

$$\alpha = P[S_t > 0], \beta = P[S_t = 0] \quad (6)$$

and

$$\alpha = P[R_t = \delta_t], \beta = P[R_t < \delta_t] \quad (7)$$

In this context, the demand δ_t is regarded as a known quantity at any time instant t . All other quantities involved, namely, S_t, X_t, R_t , and W_t , are regarded as random variables. Given the storage capacity c , the demand δ_t , and the probability distribution and autocorrelation functions of the input X_t , in theory, the probability distributions of output variables S_t, R_t , and W_t can be determined in terms of that of X_t ; this, however, is not an easy task due to the nonlinearity of the dynamics expressed in Equations 2–4. Theoretically, once the distribution function of S_t or R_t has been determined, the reliability α is obtained directly from Equation 6 or 7, respectively. However, although the theoretically based calculations unavoidably involve complete knowledge of the distribution function of S_t and R_t , the reliability α can be determined in an alternative, much simpler, manner. Under the assumptions of stationarity and ergodicity, α can be estimated from a historical (for an existing reservoir) or synthesized (via simulation) time series of storage s_t or release r_t of adequate length n . (Here lower case symbols were used for values of the random variables S_t and R_t .) Specifically, the estimate of α based on Equation 6 is

$$\alpha = \frac{1}{n} \sum_{t=1}^n [1 - U(-s_t)] \quad (8)$$

where $U(x)$ is Heaviside's unit step function, where $U(x) = 1$ for $x \geq 0$ and $U(x) = 0$ for $x < 0$. Correspondingly, the estimate of α based on Equation 7 is

$$\alpha = \frac{1}{n} \sum_{t=1}^n U(r_t - \delta_t) \quad (9)$$

The purpose of the sum in Equation 8 (or 9) is to count the periods when storage is not zero (or release r_t equals the demand δ_t). Thus, reliability is expressed as the proportion of time steps when the system performs as desired. For this reason, α has also been termed *time-based* reliability. Here, it must be observed that although Equations 5–7 are mathematically equivalent to each other, as are Equations 8 and 9, when applied to historical time series, they may result in different estimates. For example, a city's water supply may not be allowed to empty, as restrictions on releases are applied before this situation is reached. In such a case, Equation 8 may result in the erroneous estimate $\alpha = 1$, whereas Equation 9 will estimate the reliability correctly if the desired demand, before restrictions, is entered into the calculations. Thus, Equation 9 is preferable when dealing with historical time series, but both are equivalent in synthetic simulations; moreover, application of Equation 8 is faster as it does not require simulating releases at all (only Equation 2 needs to be applied).

The stationarity assumption that was inherent in the previous analysis is satisfactory when the time step is a year (either calendar or hydrologic). However, the annual time step is usually too large and hides the variation of both inflows and demand within a year, which may result in a failure some time within the year that is recovered at the end of the year. Therefore, a smaller time step (e.g., monthly) is usually chosen, so that one year corresponds to $k > 1$ (e.g., 12) time steps. On this finer timescale, all processes depend on the time step in a periodic manner, that is, they are cyclostationary. Yet, the reliability and failure probability are usually expressed on the annual scale, in which stationarity is redeemed. To shift from the finer scale to the annual scale, the rule adopted is that a failure occurring in one or more finer scale time steps is regarded as a failure for the year. Using this rule, Equation 6 becomes

$$\alpha' = P\left[\bigcap_{i=1}^k (S_{t-i} > 0)\right], \beta' = P\left[\bigcup_{i=1}^k S_{t-i} = 0\right] \quad (10)$$

where the symbols α' and β' were used instead of α and β to distinguish from time-based reliability, whereas the symbol ' $\cap$ ' indicates that all of the following events should occur simultaneously and ' $\cup$ ' indicates that any of the following events should occur. Similarly, Equation 7 becomes

$$\alpha' = P\left[\bigcap_{i=1}^k (R_{t-i} = \delta_{t-i})\right], \beta' = P\left[\bigcup_{i=1}^k R_{t-i} < \delta_{t-i}\right] \quad (11)$$

or alternatively,

$$\alpha' = P\left[\sum_{i=1}^k R_{t-i} = \sum_{i=1}^k \delta_{t-i}\right], \beta' = P\left[\sum_{i=1}^k R_{t-i} < \sum_{i=1}^k \delta_{t-i}\right] \quad (12)$$

Likewise, Equations 8 and 9 become

$$\alpha' = \frac{k}{n} \sum_{p=1}^{n/k} \min\{[1 - U(-s_t)]; t = k(p-1) + 1, \dots, kp\} \quad (13)$$

$$\alpha' = \frac{k}{n} \sum_{p=1}^{n/k} \min\{U(r_t - \delta_t); t = k(p-1) + 1, \dots, kp\} \quad (14)$$

respectively. The sums in Equations 13 and 14 count the number of years in which no failure has occurred. Apparently, α' and β' provide information on the occurrence of a failure within a year and not in the time period during which the failure lasted. Therefore, they have been known as *occurrence-based* reliability and failure probability, respectively. An overall indication of the duration of failures within an average year can be obtained by applying Equation 8 or 9 and estimating β , the time-based probability of failure.

Apart from occurrence-based and time-based reliability, an additional reliability measure has been often used, which is not expressed in terms of probability (and thus, literally does not comply with the general definition of reliability). This is the so-called *volumetric* or *quantity-based* reliability, expressed as the ratio of the average release to demand:

$$\alpha_V = 1 - \beta_V = E \left[\sum_{i=1}^k R_t \right] / \sum_{i=1}^k \delta_t \quad (15)$$

Given that a failure that occurs in a year does not extend over the whole year, and, in addition, the release during the failure is not necessarily zero but some positive quantity smaller than demand, it is easily concluded that

$$\alpha' \leq \alpha \leq \alpha_V \quad (16)$$

Among the three measures of reliability, the most important and most frequently used is the severest, the occurrence-based reliability α' . Another means for expressing virtually the same concept is the return period or recurrence interval of emptiness, T . This is the mean time between two consecutive empty states of the reservoir, and it is none other than the reciprocal of the probability of failure β' (1):

$$T := 1/\beta' = 1/(1 - \alpha') \quad (17)$$

which is expressed in years (given that α' and β' are expressed on an annual timescale). The concept of the return period of emptiness of a reservoir is similar to that typically used for design floods. The difference is that in design floods, failure is the exceedance of the magnitude of the design flood, whereas failure in a reservoir is emptying of the reservoir. Typical design values of reliability and return period for reservoir design are $\alpha' = 99\%$ ($T = 100$ years) for municipal water supply reservoirs, $\alpha' = 70 - 85\%$ ($T = 3.3 - 6.7$ years) for irrigation reservoirs in subhumid climates, and $\alpha' = 80 - 95\%$ ($T = 5 - 20$ years) for irrigation reservoirs in arid climates (2, p. 313).

TRADITIONAL RESERVOIR DESIGN PROCEDURES

Most hydraulic structures, such as flood protection works and drainage networks, whose load varies randomly, have been designed on a probabilistic basis, adopting a certain

reliability level or, equivalently, a certain return period for the design flood. Traditionally, however, this has not been the case in reservoir design, which has rarely been based on sound probability. This is obvious even from the terminology traditionally used. For example, the use of the term *firm yield* implies a nonprobabilistic, or failure-free concept. Specifically, the firm yield of a reservoir has been defined as the draft or withdrawal that lowers the water content in a reservoir from a full condition to its minimum allowable level just once during a critical historical drought. (3, p. 27.8; 4, p. 534). It has been characterized as essentially the no-failure yield (3, p. 27.8). In a probabilistic context, however, any draft has a nonzero probability of failure (unless the demand is less than the hypothetical lower bound of the inflow distribution, which can be plausible only for perennial streams; this is unusual).

Several procedures have been widely used in reservoir design, which are rather deterministic and not consistent with the reliability concept. The most common has been *mass curve analysis* and its variations. A mass curve is a plot of cumulative inflow volumes (typically based on historical discharge records) as a function of time. Using this plot, the firm yield, as well as the required reservoir storage to attain this firm yield, can be determined graphically. In addition, the method can determine the required storage for a smaller target release. This graphical method was developed 120 years ago (5) and has been widely used until now, although criticized (6) for not providing information on the probability of failure and for the fact that the reservoir capacity determined by this method increases as the arbitrary length of available observed inflow data increases. As shown by Feller (7), this increase is asymptotically proportional to the square root of the length of the record.

A first variation of the method is its application using synthetic, rather than observed, data (6). This eliminates the drawback of the arbitrary length of the record and also provides some measure of uncertainty by applying the same procedure using different generated synthetic series. However, this kind of description of uncertainty is not consistent with a rational definition of reliability (e.g., that of the previous section).

A more theoretical flavor for the method has been given by the so-called *range analysis*, commenced by the work of Hurst (8; see also 9, p. 184). Range is essentially the algebraic difference of the maximum and minimum departures of the mass curve from the straight line that joins its starting and ending points. The range concept has greatly contributed to the understanding and description of the so-called Hurst phenomenon in hydrology, climatology, and other geophysical sciences. Applied to a reservoir, the range represents the required storage of a reservoir operating without any spill or other loss and providing a constant outflow equal to the mean flow. Obviously, this is an oversimplification of a real reservoir. On the other hand, the range concept involves complexity in estimation, and simpler and more efficient methods have been proposed that can be used instead of range analysis (10).

An additional design method is the so-called *sequent-peak analysis* (11, p. 274; 12, p. 400). Essentially, it is

a tabulated version of mass-curve analysis and can also incorporate in the calculations, apart from runoff, the effects of precipitation, evaporation, and leakage. The method does not involve the reliability concept, nor does it consider spills from the reservoir.

SIMPLIFIED RELIABILITY-BASED PROCEDURES FOR RESERVOIR DESIGN

As already mentioned, the analytical determination of reliability in the general case of a reservoir fed by inflows with seasonality, that have arbitrary probability distribution, and autocorrelation functions is a very difficult, if not impossible, task (1). Therefore, existing analyses have been based on several simplifications. However, the results of such analyses are very useful, at least for the initial stage of reservoir design. The typical simplifying assumptions are to

- neglect secondary inflows (precipitation) and losses (evaporation, leakage);
- neglect seasonality by the adoption of an annual time step;
- neglect autocorrelation and assume that inflows are independent in time;
- use a specific distribution function for inflows, typically two-parameter such as normal, lognormal, or gamma.

The objective of such probability-based theoretical analyses is to determine the relation of the following three quantities:

- reservoir size c ; it is usually standardized as $\kappa := c/\sigma$, where σ is the standard deviation of annual net inflow X_t ;
- demand δ , which is assumed constant for all years; it is usually standardized as $\varepsilon := (\mu - \delta)/\sigma$, where μ is the mean of annual net inflow X_t ; ε has been termed the standardized inflow (8; 3, p. 27.7) or the *drift* (1);
- probability, expressed either as reliability α , probability of failure β , or return period T ; because of the annual time step used, time-based reliability (Equations 6, 7) is identical to occurrence-based reliability (Equations 10–12).

The first among the probabilistic approaches used in such analyses is discretization of reservoir storage into several zones, each representing a certain state, and the use of a Markov chain model to represent transitions from state to state (13,14; see also 9, p. 264).

A second method is stochastic (Monte Carlo) simulation, in which a long synthetic series of inflows is generated from the appropriate distribution function and then transformed into a series of storage values using Equation 2; reliability is then easily determined from the storage time series using Equation 8. Gould (15) used this method to propose a reservoir size–yield–reliability formula, fitted to 240 sets of Monte Carlo simulations

for various combinations of demand, reservoir size, and skewness of gamma distributed inflows:

$$(\varepsilon + 0.15)[\kappa + d_1(\alpha, \gamma)] = d_2(\alpha, \gamma) \quad (18)$$

where d_1 and d_2 are coefficients depending on reliability α and skewness γ and are given by nomographs (see also 2, p. 323). McMahon and Mein (16) adapted this formula to indicate reliability more explicitly; this can be estimated from the standardized normal variate z_α corresponding to α , using the equation

$$z_\alpha = 2\sqrt{\varepsilon(\kappa + d(\alpha)\sigma/\mu)} \quad (19)$$

where d is a coefficient depending on reliability α and is given by a table (see also 3, p. 27.14).

A more accurate and rigorous theoretical methodology to estimate the reservoir size–yield–reliability relationship was developed by Pegram (1). This was based on finite-difference and integral equations, which employ reservoir dynamics (Equation 2) in a probabilistic context to determine the return period of emptiness. Pegram applied his methodology for normal, lognormal, and discrete inputs both independent and serially correlated. His results, when compared to those of the Gould method (Equations 18–19) indicate that the latter underestimates the reservoir size required to attain a certain reliability level. Using Pegram's results for normally distributed inflows, which were verified and expanded here with extended simulations, the following approximate relationship has been established:

$$\ln(T - 1) = 2(\varepsilon + 0.25)(\kappa + 0.5)^{0.8} \quad (20)$$

This is valid for $T > 2(\alpha > 0.5)$ and can be alternatively written as

$$\begin{aligned} \ln(T - 1) &= -\ln(1/\alpha - 1) \\ &= (2/\sigma^{1.8})(\mu + 0.25\sigma - \delta)(c + 0.5\sigma)^{0.8} \end{aligned} \quad (21)$$

For known mean μ and standard deviation σ of inflows, Equation 21 can directly yield either the reliability α for a known reservoir size and demand, the reservoir size c for a given demand and reliability, or the demand δ that can be met at a given reliability for a known reservoir size. Equation 20 is graphically depicted in Fig. 1 in comparison with Pegram's exact results. This equation is suggested for preliminary estimates, but it should be applied with caution for the reasons explained in the next section.

Effects of Inflow Characteristics on Reservoir Size

As explained earlier, simplified design procedures such as that using Equation 20 are based on a number of abridging assumptions about inflows. Significant differences may appear if these assumptions are not valid. More specifically, what may cause significant departures from Equation 20 are hydrologic persistence, especially long-term, and the seasonal distribution of inflow and demand. Less significant differences are caused by the skewness of inflows and secondary inflows and losses.

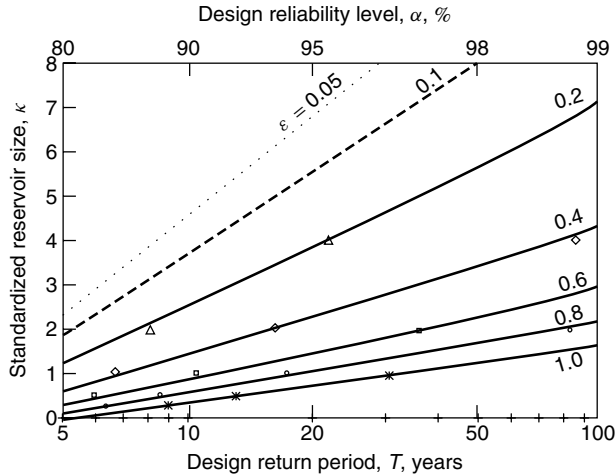


Figure 1. A simple representation of reservoir size–yield–reliability: Standardized reservoir size ($\kappa = K/\sigma$) required to achieve a certain drift ($\varepsilon = (\mu - \delta)/\sigma$) at a certain reliability level for independent inputs normally distributed. Lines are constructed from Equation 20, whereas plotted points are theoretical results by Pegram (1) for $\varepsilon = 0.2$ (triangles), 0.4 (diamonds), 0.6 (squares), 0.8 (circles), and 1.0 (stars).

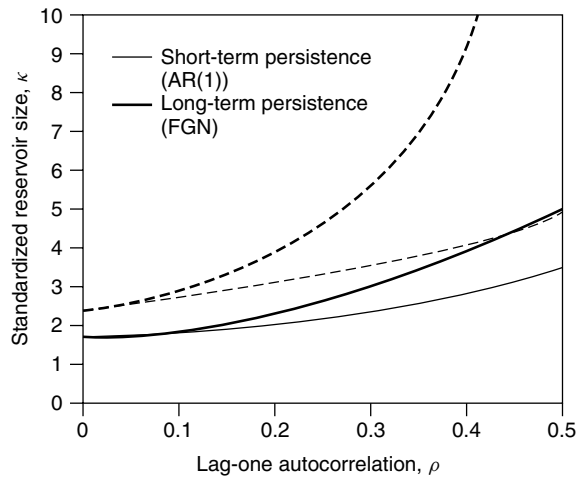


Figure 2. Effects of short-term and long-term hydrologic persistence of inflows on the standardized reservoir size κ required to achieve a drift $\varepsilon = 0.8$ at a reliability level $\alpha = 98\%$ (continuous lines) and a drift $\varepsilon = 0.2$ at a reliability level $\alpha = 90\%$ (dashed lines). Results are obtained by simulation.

The effect of hydrologic persistence is demonstrated in Fig. 2, which depicts the standardized reservoir size κ required to achieve two combinations of drift and reliability ($\varepsilon = 0.8, \alpha = 98\%$ and $\varepsilon = 0.2, \alpha = 90\%$) versus the lag-one autocorrelation coefficient, ρ . Two cases of hydrologic persistence have been examined, short-term and long-term. In short-term persistence, it was assumed that the inflows follow the autoregressive process of order 1 [AR(1) or Markov], whereas in long-term persistence, it was assumed that they follow the fractional Gaussian noise (FGN) process with Hurst exponent $H = \ln(2 + 2\rho)/\ln 4$. Obviously, the effect of persistence is very significant, especially for long-term persistence and high

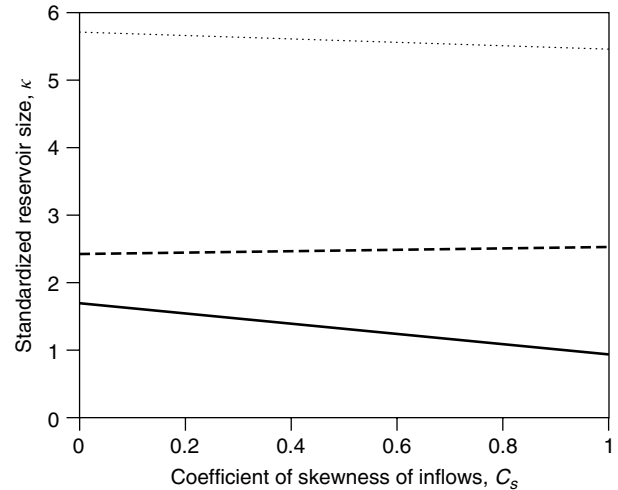


Figure 3. Effect of the skewness of inflows on the standardized reservoir size κ required to achieve a drift $\varepsilon = 0.8$ at a reliability level $\alpha = 98\%$ (continuous line), a drift $\varepsilon = 0.2$ at a reliability level $\alpha = 90\%$ (dashed line), and a drift $\varepsilon = 0.2$ at a reliability level $\alpha = 98\%$ (dotted line). Results are obtained by simulation using independent two-parameter gamma distributed inflows.

demand (low drift ε). Thus, the required reservoir size for $\varepsilon = 0.2$ and $\alpha = 90\%$ is $c = 2.4\sigma$ when $\rho = 0$ ($H = 0.5$) and becomes 4 times larger ($c = 9.6\sigma$) when $\rho = 0.4$ ($H = 0.74$).

To quantify the effect of seasonal variation of inflow and demand, it is observed that in the worst case, the total annual inflow comes before the beginning of withdrawal (the “sudden” model). The reservoir dynamics in Equation 2 assumes that inflow and withdrawal are distributed evenly during a year (the “steady” model). If inflow precedes withdrawal, Equation 2 should be modified to read $S_t = \max[0, \min(S_{t-1} + X_t, c) - \delta_t]$. It is then easily determined that extra storage equal to δ is required in addition to that estimated from Equation 21. This worst case, however, is not very realistic, except in arid regions. In the real world, the extra storage capacity required (in addition to that estimated from Equation 21) is a percentage of δ . This can be as high as 50% for water supply reservoirs and 80% for irrigation reservoirs in semiarid regions.

The effect of the skewness of inflows is demonstrated in Fig. 3, which depicts the standardized reservoir size κ required to achieve three combinations of drift and reliability ($\varepsilon = 0.8, \alpha = 98\%$; $\varepsilon = 0.2, \alpha = 90\%$; and $\varepsilon = 0.2, \alpha = 98\%$) versus the coefficient of skewness, C_s . It can be observed that the effect of skewness is not significant; for low draft (high drift ε), this effect can be beneficial (lowering of required storage), but for high draft, the required storage is practically insensitive to skewness.

GENERALIZED SIMULATION PROCEDURE FOR RELIABILITY-BASED RESERVOIR DESIGN

The simplest general procedure for estimating the reservoir size–yield–reliability relationship in an accurate and detailed manner and for any arbitrary inflow characteristics is stochastic (Monte Carlo) simulation. A simplified simulation procedure is outlined as follows:

1. Generate a series of inflows X_t at an appropriate timescale (e.g., monthly) using an appropriate stochastic model (e.g., 17, 18; see also the entry STOCHASTIC SIMULATION OF HYDROSYSTEMS).
2. Assume a reservoir size c .
3. Calculate a series of reservoir storages using Equation 2.
4. Estimate the reliability using Equation 13.
5. Repeat steps 3–4 for different reservoir sizes.

This procedure is very simple to execute even in a tabulated form on a spreadsheet.

The drawback of stochastic simulation is that it requires a vast simulation length to find accurate results. It can be shown that the required number of simulated time steps (e.g., months) to estimate the occurrence-based failure probability β' at an acceptable error $\pm \varepsilon \beta'$ and confidence γ is

$$n = k(z_{(1+\gamma)/2}/\varepsilon)^2(1/\beta' - 1) \quad (22)$$

where k is the number of time steps per year and z_p is the p -quantile of the standard normal distribution. For instance, for $k = 12$, $\gamma = 95\%$ ($z_{(1+\gamma)/2} = 1.96$), $\varepsilon = 10\%$, and $\beta' = 0.01$, this yields $n = 456,000$ months (38,000 years). Today, this is not a major problem as the required computer time for such a simulation length can be less than one second in a common PC.

The simplified procedure described can be extended to a detailed simulation procedure, which includes, in addition to runoff, the precipitation, evaporation, and leakage of the reservoir. In this case, level–area–volume and level–leakage relationships are required to establish the functions $a(S)$ and $l(S)$ which yield reservoir area a and leakage l for any storage S (usually using interpolation from arrays of tabulated values). In this case, the runoff Q , has to be expressed in equivalent depth units, as are precipitation P and evaporation E . If f is the catchment area, the net inflow becomes

$$X_t = Q_t[f - a(S_{t-1})] + (P_t - E_t)a(S_{t-1}) - l(S_{t-1}) \quad (23)$$

where it was implicitly assumed that variations in the reservoir area and leakage within a time step are not large, so that $a(S_{t-1})$ and $l(S_{t-1})$ can be assumed representative of the entire period within a time step. When the detailed simulation procedure is employed, three time series, instead of one, have to be synthesized. Obviously, Q_t and P_t are cross-correlated, and therefore they cannot be generated independently of each other; a bivariate stochastic model is needed in this case. On the other hand, E_t can be generated independently of the other two series.

Obviously, the simulation procedure, either simplified or detailed, can be applied directly using historical, rather than synthesized inputs. However, due to short record length, the accuracy of results will not be satisfactory.

Extension of the simulation method, combined with optimization, for a multiple reservoir system can be found in References 19 and 20.

DESIGN OF ADDITIONAL STORAGE ZONES OF A RESERVOIR

The design of flood control storage is typically reliability-based but in a very different context from that described for active storage. A failure of the flood routing function of a reservoir is not a performance failure but a structural one: an overtopping of the dam due to a severe flood can result in collapse of the dam. Therefore much lower levels of probability of failure are adopted, of the order of $10^{-3} - 10^{-6}$. The typical steps here are (1) estimation of a design storm, based on statistical analysis of rainfall, for an appropriate return period ($10^3 - 10^6$ years); (2) estimation of the inflow hydrograph using an appropriate rainfall–runoff model; and (3) routing this hydrograph through the spillway and estimation of the outflow hydrograph and the maximum water level. Implicit assumptions in the entire procedure, like the assumption that the reservoir is full at the beginning of the flood, decrease the risk further. It must be noted that several procedures have been proposed that are supposedly risk-free. These are based on the so-called probable maximum precipitation concept. However, it has been argued that a risk-free procedure is an illusion and the value of probable maximum precipitation can be exceeded by a certain probability (e.g., of the order of 10^{-5} ; 21).

The sizing of dead storage has been typically based on the expected sediment accumulation in the reservoir for a certain design period (e.g., of the order of 10^2 years). Because of the large design period and the accumulative character of this process, the approach to this problem is very different. Expected values, rather than probabilities, are involved in the calculations. The additional objectives that the dead storage serves, environmental protection (protection of the habitat of the reservoir during dry periods) and conservation of the quality of the landscape, have not been given special attention, until now, and have not been considered in the design procedure. One would expect that, some years after construction, these additional objectives would not be served adequately because the water in the dead volume would be reduced due to sediment accumulation. Fortunately, however, this has not been the case: the implicit assumption that sediments will reach the bottom of reservoir near the dam is not verified. Thus, in large reservoirs, sediment accumulation occurs mainly in the active zone of reservoir (near the entrance of the river to the reservoir) and much less in dead storage. The unfavorable consequence is the reduction of active storage. These problems need to be further investigated in future reservoir designs.

Acknowledgment

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LAKES

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Lakes are waterbodies of considerable size that are larger than ponds. *Shallow lakes* are mixed by the wind from the surface to the ground, and the daylight reaches

the bottom. *Stratified lakes* are deep enough to be vertically divided into a lighted surface-near *euphotic* zone, where photosynthesis of green plants is possible, and a deep-water *aphotic* zone, which is too dark for photosynthetic activities. In many lakes, thermal vertical stratification is observed as a warm layer near the surface, the epilimnion, and a colder layer below, the hypolimnion. The epilimnion is wind-exposed and well mixed, whereas the deep water layer of the hypolimnion is not included in the mixing process. Introductions to the ecology, hydrology, and limnology of lakes are available in textbooks, handbooks (1,2), and reviews (3–9).

ORIGIN OF LAKE BASINS

Lake basins containing standing waterbodies were formed by different processes; tectonic forces; volcanic, glacial, fluvial activities; or by solution of easily dissolvable rock such as limestone. Hutchinson (1) ordered the lakes by their mode of origin into about 80 categories. Typical examples are listed by Meybeck (8,10), who also gives numbers of the types of origin, of size, and of age.

Tectonic lakes are found in geological rift valleys; the deepest are Lake Baikal (11) and Tanganyika (12). The Caspian Sea and the Aral Sea (13) are rest areas of the ancient Thetis Sea which was closed by tectonic movements, and they are the largest inland waters in area (374,000 km², 64,500 km²). The oldest inland waters are found among tectonic lakes, some of these, it is estimated, are older than 20 to 30 million years.

Volcanic processes led to the origin of different lake types. *Maar lakes* emerged after singular volcanic events (steam explosions in contact zones of magma and groundwater). Lava dams or emerging volcanoes resulted in reservoir-like lakes, such as L. Kivu and Laguna del Laja (Chile). Large lakes were formed on lava plateaus (L. Myvatn, Iceland) and in *caldera* basins, whereas *crater lakes* are usually small. Lakes at active volcanoes are often heavily acidified by acidic brines rinsing into the lake water (14).

The main types of glacially formed lakes are (1) small *kar lakes* at the slopes of mountains at different altitudes, (2) *piedmont lakes* where glaciers from valleys reached the foreland at the foot of the mountains, and (3) extended *lake districts from low altitude glaciations* as on the Canadian Shield and the circum-Baltic area. Postglacial lakes of temperate climatic zones are only 8 to 12 thousand years old.

In the lowland area of large rivers, the flow of the running rivers often changes, leaving standing waters, oxbows or side waters, as *fluvial lakes*, which were parts of the rivers in the past or are permanently in open connection with the river system.

Lakes are more numerous as size decreases. Meybeck (8,10) gives the estimated worldwide number of lakes, ordered by size classes and by the origin of lake basins. A list of the deepest and largest lakes, by area and by water volume, is given by Herdendorf (15,16), including many uncertainties on the basic data of even the 50 most important inland waters of the world.

WATER BUDGET AND SALINITY OF LAKES

The oceans contain 83.5% of the world's water, the inland waters only 0.015%. Lakes contain a total water volume of 205,000 km³; about 50% of this total volume is freshwater, on the one side, and saline water on the other side. The deepest and oldest lake, Baikal, has a volume of about 20% of the total freshwater volume of lakes and is the worldwide largest singular freshwater resource. The annual fluxes through the pools of the atmosphere, rivers, and groundwater are 496, 38, and 12 thousand km³; exchange times are 10 days, 16 days, and 700 years, respectively. The volume of freshwater lakes is 100 thousand km³; exchange times are 1 month to 500 years [data after (17,18)].

Lakes are standing waters that are usually connected with and included in the flow system of river catchments. The lakes within the catchment systems can be ordered by their degree of connection to the overall water flows (19). Lakes with intensive through-flow have high rates of flushing and water exchange and short retention times. The lake volume compared with the volume of all inflows per year gives the virtual *filling time*. The relation of the outflow and the lake volume gives the net exchange rate of the lake water. Including groundwater exchange, precipitation on the lake surface, and evaporation, the relation of all inputs to all losses of water gives the gross exchange rate; its reciprocal value is the *retention time* or average age of the water.

Depending on the given regional water balance, lake basins are permanently flushed, have an outflow, are in a steady state of the flow of water and of dissolved salts, and, thereby, contain freshwater. In areas with a negative water balance, the river catchments are *endorheic* systems, and terminal lake basins occur without outflows; here, the water finally evaporates, and the inflowing salts accumulate, leaving *saline lakes*. Saline lakes are typical of (semi)arid areas and contain nearly half of the world's lake water volume (20). Saline lakes (with salinities >3‰) (21,22) are found in all climatic zones, warm, temperate, and cold regions, where negative water balances are given by large-scale climatic zones or by regional orographic conditions (rain-lee east of the Rocky Mountains in North America, east of the Andes in South America, or in the Iranian basin). Such endorheic river catchments, in which the water flow ends in terminal lakes, are the Jordan River system with the Dead Sea, the system of Lake Titicaca with Lake Poopó as the terminal saline lake, and the system of Lake Tahoe and Pyramid Lake in North America. In Central Australia, Lake Eyre is a large, endorheic saline basin. The highest possible salt concentrations are found in the Dead Sea (~300 g/L) and in Deep Lake (Antarctica) which never freezes; the water reaches -14.5 °C during the winter (22).

STANDARD COMPOSITION OF FRESHWATERS

Freshwaters, soft waters as well as hard waters, are chemically dominated by the carbonate system, CO₃²⁻ and HCO₃⁻ as anions, and Ca²⁺ and Mg²⁺ as cations; together they are the main constituents and the buffering

system of neutral freshwaters. This is caused by rainwater that is weakly acidic from CO₂ and, thereby, dissolves the carbonatic minerals of soils and rock (23,24). In areas poor in carbonates, the water contains low concentrations of dissolved salts, and in other areas of dominating limestone, the water is rich in dissolved carbonates. The qualitatively similar composition of freshwaters is described as a worldwide "standard water composition" (25,26).

The gases in air, 78.09% N₂, 20.95% O₂, and 0.03% CO₂, are in physical equilibrium between air and water. At low temperatures, higher concentrations of gases dissolve in the water than at high temperature. The maximum content of dissolved oxygen and CO₂ in water is 14.5 (8.9) mg O₂/L and 1.005 (0.51) mg CO₂/L at 0 °C (20 °C).

STRATIFICATION AND MIXING

Lakes are usually stratified into an upper layer, the epilimnion, which is well mixed by the wind, and a deep waterbody, the hypolimnion, which is not included in the mixing process. Stratification occurs seasonally in the temperate climatic zones, or in a diurnal-nocturnal rhythm in the tropical zones (Fig. 1). Ice covers block energy inputs by wind into the water column during the winter. The regime of stratification and mixing of the water column, as given by the climate, defines the types of lakes as (1) *monomictic*, (2) *dimictic*, or (3) *polymictic*, according to the number of periods of total mixing (*holomixis*) per year. Tropical lakes at low altitude are polymictic; they mix during the night and stratify during the day. Dimictic lakes have a stable stratification with a warm epilimnion during the summer and an ice cover during the winter; they show holomixis during the two seasonal periods of homothermy. One holomixis occurs in spring after ice melting when the surface water has the same temperature as the hypolimnion, usually 4 °C, and a second during late autumn before freezing. Monomictic lakes mix only once per year. The warm-monomictic lakes of oceanic-temperate regions never freeze and show holomixis during the winter. Cold-monomictic lakes are found in polar climatic regions; they are covered by ice during large parts of the year and melt only in summer when the water column reaches 4 °C and mixes from the surface to the ground.

The stability of stratification of a warm epilimnion and a cold hypolimnion determines whether wind forces mix only the uppermost layers, or the entire water column from the surface to the bottom. Stratification stability depends on the density gradient between the epilimnetic and the hypolimnetic partial waterbodies. Indicative numbers are the *Wedderburn Number* or the *Lake Number*, which, at given threshold values, predict optional holomixis with the next wind-forcing (27).

Climatic Zones and Lake Temperatures

Lakes are integrators of the regional climate and weather development (28). The surface temperatures follow the actual air temperature with a time delay, and lakes of the same region show time courses of similar surface temperatures. The deep water of lakes in polar and

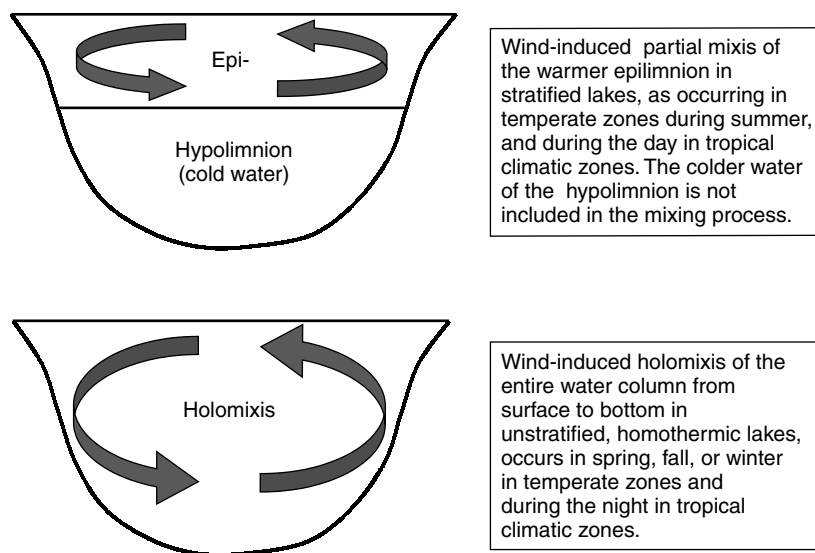


Figure 1. Seasonal or diurnal succession of stratification into warm epilimnion and cold hypolimnion and unstratified water column when the water is homothermic with complete overturn of the water from surface to ground (holomixis).

temperate continental climatic zones is at 4°C, the temperature of water at its maximum density. The deep water of warm-monomictic lakes of temperate oceanic climatic zones in the Southern Hemisphere, in New Zealand and South America and also in western and southern Europe, has a temperature above 4°C. The hypolimnetic temperatures remain from the last holomixis at the end of winter, and changes reflect long-term climatic conditions (29).

Waves and Oscillations

Driven by wind, waves emerge on the surfaces of lakes; their heights can be approximated by $h = 0.105 (\text{fetch})^{0.5}$ (wave height and fetch given in cm; the *fetch* is the maximum length of the wind blowing across an open lake surface) (9). Less peculiar are oscillations along the lake axis, which have amplitudes of few centimeters at the lake surface (*seiches*). Their frequencies depend on the length of the lake axis, between minutes in small lakes and one day in large lakes. At the boundary layer between the epi- and hypolimnion, however, internal oscillations can reach wave amplitudes of 10 m or more. The periods of these internal oscillations are slower and reach 1 month in the largest lakes, such as Baikal or Tanganyika (1,27).

ECOSYSTEMS OF LAKES

The ecosystems of lakes belong to the best described and understood ecosystems because the composition of the community is limited in number of species and the populations are limited by the lake size. The seasonal development of the populations, their change in species composition, and the growth and decrease of the populations are well investigated and analyzed in terms of numbers of individuals as well as in terms of energetics and species composition (30).

Lakes have characteristic compartments as partial habitats of their ecosystems. The ecosystem of the pelagic zone of lakes is dominated by planktonic microorganisms,

larger zooplankton, and fish. The phytoplankton is autotrophic and limited to the euphotic layers near the surface where light intensities are sufficient for photosynthesis and oxygen production. The limit between the *euphotic* and the *aphotic zone* is usually at depths where the light intensity decreases to <1% of that at the lake surface.

The *pelagic zone* is the body of free water of the lake, the *littoral zone* is the euphotic part at the lakeshore, and the *profundal* is the aphotic dark layer of the *benthic zone* below the littoral. The benthic community outside the littoral zone, the profundal, consists of the organisms living in and on the sediments, microorganisms, macrozoobenthos, and benthic fish species. Below the light limit for photosynthesis, the species of the benthic community, macro- and microorganisms, are heterotrophic and consumers of oxygen. All particles in the water eventually sink to the lake bottom, form annual layers of growing sediments (*varves*), and leave an archive of residuals. Therefore, the history and the development of the lake can be reconstructed from the microlayers of a sediment core, with respect to eutrophication and nutrient levels and impacts of climatic changes.

The littoral zone is the euphotic part of the benthic zone. The living community consists of autotrophic macrophytes and algae and of heterotrophic animals and microorganisms. The belt of macrophytes appears as an interface between land and water; wetland plants are on the land side of the belt, followed by plants rooting in the water and emerging in the air, swimming plants, and lastly submerged plants that cover the light bottom of the deeper littoral zone. The partial ecosystem of the littoral zone connects the land and water, and simultaneously, as a biologically reactive zone, retains and changes the flow of nutrients and other allochthonous inputs of matter into the lake's pelagic zone.

Trophic State and Productivity

Trophy is a measure of the level of nutrients and of the biological productivity resulting from nutrients supplies.

The organismic productivity of aquatic ecosystems, measurable as the formation of biomass, is limited by the availability of the rarest nutrient element (C,N,P,S,Fe,...). In temperate zones, phosphorus is usually the least available element because there is no gaseous transport form through the air and P is tightly sorbed on soil particles and retained on the land. The production of organic substance is limited by the availability of phosphorus, so the trophy (productivity) of lakes can be related to the P-content of the water at the start of the growing season (31,32). The process of growth needs a proportion of 106C:16N:1P as the atomic ratio of biomass (33,34). The primary production of phytoplankton and macrophytes—depending on P—reaches annual rates of 10 to 50 g C/m² of organic carbon in nutrient-poor lakes and more than 500 g C/m² in very nutrient-rich systems.

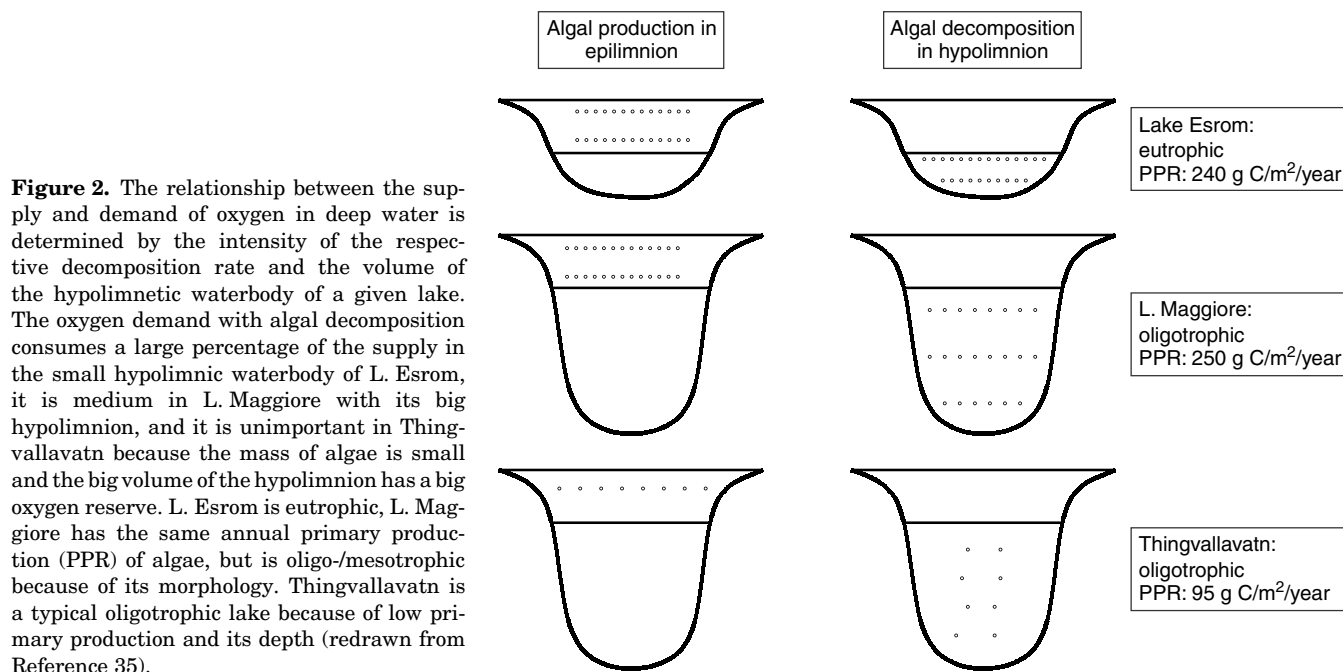
The biomass produced sediments to the lake bottom, where this dead organic substance is decayed by oxygen-consuming bacteria. Depending on the trophic state, the oxygen conditions in the hypolimnetic water body reach different end points in late summer and fall. In oligotrophic lakes, the nutrients and productivity are low, as well as the following decomposition and oxygen losses in deep water. In eutrophic lakes, however, the nutrient supply and the resulting productivity are high. After decomposition in deep water layers, the oxygen content in the hypolimnion may reach critically low limits in late summer. The consequences of high nutrient supplies are mitigated in deep lakes with high hypolimnetic volumes but are more severe in shallower lakes with small hypolimnetic volumes and small oxygen reserves therein (Fig. 2).

Food Chains, Filtrators, and Size Spectra in plankton

Within the pelagic zone of lakes, the species of the plankton community can be structured by the food chain from the primary producers, the green algae, to

herbivores, zooplankton species that feed on these algae, and carnivores that feed on herbivorous zooplankton. The efficiency of energy transfer from the lower to the higher trophic levels is about 10% from plants to animals and up to 30% from animal prey to their predators. Because there are losses of unused particles and dissolved organic matter (DOM) on all levels, recycling by bacteria as decomposers of DOM is important. Based on the bacteria as food, leading via bacteria-feeding protozoans to larger predators, the “lost” DOM is recycled again into the “normal” food chain by the “microbial loop.” This secondary food chain is important for the overall efficiency of the ecosystem as a processor of energy and its ability to recycle nutrient elements.

In oligotrophic systems, the organisms, prey and predators, are present in low concentrations, and a predator has to find and to grasp each single prey individually. The prey in such food chains is usually smaller than its predator by a factor of 10 to 20 by body length. Therefore, in oligotrophic planktonic systems, the food chain from green algae, via zooplankton to fish needs four to five subsequent trophic levels. The overall transfer efficiency with its losses at any transfer is small. In the more productive systems of meso- to eutrophic lakes, however, the concentrations of organisms from algae and bacteria to zooplankton and fish are high, and additional, new types of filtering predators appear. Filtrators use a filtering system, comb-like sieves in cladocerans (*Daphnia*), fine gill structures in filter-feeding fish (Coregonids), to collect large numbers of very small prey particles. The linear size relation between prey and filter-feeding predators is 100 × to 1000 ×, and, thereby, the length of the food chain becomes short, three levels from algae to fish. In temperate lakes, we find *Daphnia* as the keystone filtrator on the level of herbivores all over the world. The filtering, zooplankton-feeding fish of the genus



Coregonus are found only in the Northern Hemisphere. In the ancient Lake Tanganyika, an evolution of a “new” filtrator occurred, *Limnotrissa*, where a zooplankton-feeding sardine, led to very high fish production. In other ancient tropical lakes, herbivorous *Tilapia* species appeared that filter-feed on algae.

The food web of any ecosystem might be described on the level of species composition and populations, according to its taxonomic structure. The system, however, might be alternatively viewed with an ataxonomic approach and explained functionally as a system of trophic levels linked by energy transfers: the “trophic-dynamic system” (34,36).

Another way to describe the planktonic food web and to explain some of its functions and properties is to view its size structure and the continuity of sizes as a spectrum. Investigations of the entire planktonic system revealed the plankton as a spectrum of size classes, in which the biomass of all size classes is equal. Many small organisms are present with the same sum of biomass, as are the fewer organisms of larger sizes. The regular pattern of body sizes over all organisms of plankton (37,38) corresponds to growth and physiological rates in a reciprocal way, fast overturn in the pools of small organisms and slow overturn in the big ones.

REGIONAL LAKE LIMNOLOGY

Glacial Lakes

The glaciated areas of the Canadian Shield, the Scandinavian mountains, and the adjacent sedimentary platforms

of the circum-Baltic countries have been eroded by glaciation that left ten thousands of large and small lake basins. The large basins of the Hudson Bay and the Baltic Sea are corresponding central basins. The Baltic Sea was intermittently an inland lake and then became a brackish shelf sea. The surrounding areas from Finland to northern Germany have extended lake districts that cover large parts of the land. The “limnic ratio” is a measure of the percentage of the recent lake area of the deglaciated land area (8): world total deglaciated area: 6.8%; Scandinavian Shield: 12.2%; Finland: 9.4%; Norway: 13.9%; Sweden: 8.55%; South Baltic Platform: 2.2%; Canadian Shield: 10.3%; and Patagonia: 3.0%. The lakes of this origin are the largest in Europe, Lakes Ladoga and Onega in western Russia and the large southern Swedish lakes, and in North America, the Laurentian Great Lakes and the large lakes of Canada (Table 1).

Formerly glaciated mountain districts are found on all continents (Table 1). The fjord lakes and foothill lakes in the deglaciated mountain areas are found in the Northern and the Southern Hemispheres of the temperate climatic regions in Scandinavia, the loughs/lochs of Ireland and Scotland, the English Lake District, the Alps, New Zealand, the Patagonian mountain areas of Chile and Argentina, and the Central and Northern Asian mountains. *Fjord lakes* are the deepest glacial lakes. Their mountain valley erosion basins have peculiarly steep borders and flat bottoms with depths of 500 m or more. Where the glacial erosions from valleys extend into the

Table 1. Large Glacial Lakes on Shields, Sedimentary Platforms, and in Mountain Areas

Glacial Lakes	Surface Area, km ²	Max. Depth, m	Water Volume, km ³	Retention Time Years	Age Origin, 10 ³ y	Data Source
<i>Canadian Shield lakes</i>						(8,10,49)
Superior	82,100	405	12,200	180	8, glacial	
Huron	59,500	223	3,500	48	8, glacial	
Michigan	57,750	281	4,800	31	8, glacial	
Great Bear	31,326	446	2,200		glacial	
Great Slave	28,568	614	2,088		tectonic, glacial	
<i>Andean lakes</i>						(39,40)
Argentino	1,410	500				
Gral. Carrera, Chile/Buenos Aires, Arg.	1,892	586	>500		tectonic, glacial	
Nahuel Huapi, Arg.		464				
Llanquihue, Chile	871	317	159	74		
<i>New Zealand</i>						(74)
Taupo	616	165	59	10.6	volcanic, glacial	
Waikitipu	289	380				
Wanaka	180	311				
<i>Baltic lakes</i>						(8,10,72)
Ladoga, Russia	18,130		908		>8, baltic	
Onega, Russia	9,700		292		>8, baltic	
Vänern, Sweden	5,648		152	8.8	6.5, baltic	
Vättern, Sweden	1,912	128	74	58	6.5, baltic	
Mälaren, Sweden	1,096		14	2.3	6.5, baltic	
<i>Alpine piedmont lakes</i>						(28,54)
Geneva, W.-Alps	580	310	89.9		piedmont	
Constance, N.-Alps	472	253	47.7	4.15	piedmont	
Garda, S.-Alps	370	346	49	26.6	piedmont	
Maggiore, S.-Alps	213	370	37.5	4.1	piedmont	
Como, S.-Alps	146	410	22.5	4.5	piedmont	

foothill areas of the foreland, the characteristic *piedmont lakes* cover large areas, but they are shallow compared with fjord lakes or fjord-like parts of the same lake (e.g., L. Garda, L. Argentino, L. Buenos Aires, show both, deep fjord-like parts and large, shallow foothill areas).

Lakes in the (sub-)polar regions are covered by ice permanently or during the largest part of the year. Lake Vanda (Antarctica) has a permanent, clear, ice cover. The deepest water layers are stabilized by saline inputs and trap the down-welling light energy, leading to a temperature of 25 °C above the bottom (63 m) (22). Some large lakes in the Antarctic were found by satellite radar imaging under several thousand meters of glacial ice cover. The largest of these lakes, L. Vostok, is the size of Lake Ontario (39).

TROPICAL AND ANCIENT LAKES

The lakes of the (sub-)tropical climatic zone are described in textbooks by Beadle (42) and Serruya (43). The most important difference from temperate lakes is the diurnal variability of temperatures that are as large as the seasonal variability between summer and winter in temperate lakes. The restricted time of nocturnal holomixis presumably limits the maximum depth of mixis in deep tropical lakes. Therefore, the very deep lakes Tanganyika and Malawi are meromictic, and, consequently, have anoxic, deep waterbodies. The deep African Rift Valley Lakes are ancient lakes and, therefore, have peculiarities beyond their tropical sites (12).

The largest and deepest lakes of the world are also the oldest lakes: Baikal, Tanganyika, Malawi, and Issik-Kul. The most important ancient lakes are listed in Table 2. The old lakes originate from the Tertiary and have a long development with changing water levels that could be reconstructed by paleolimnological investigations. The best investigated ancient lake is L. Biwa, whose history is based on sediment cores that are 200 and 1000 m

long, covering 0.5 to 1 million years (44). The oldest and deepest of the ancient lakes gave time to their biota for evolutionary processes which led to adaptive radiations of—mostly endemic—species, found 10 to 100× more numerous than in young lakes. There are >400 species of Cichlid fish species in Tanganyika and Malawi and >200 species of Gammarid crustacean species in Baikal. The ecosystem structure is severely changed by new keystone species in the food web and new top carnivores: in Lake Baikal the pelagic Gammarid *Macrohectopus* and Cottoidei fish and the freshwater seal *Phoca sibirica*. In Lake Tanganyika, the species *Limnotrissa* directs the food chain to high production of fish, whereas in the other large African lakes the insect larva *Chaoborus* plays the role of the main zooplankton feeder. This species leaves the lake when it becomes an adult insect, so the branch of the food chain is then lost for the lake ecosystem.

MAN-MADE LAKES: RESERVOIRS AND PIT LAKES

Man-made lakes are built in great numbers, ten thousands of reservoirs and pit lakes in the voids of surface mining. They show limnological peculiarities that come primarily from the management regime for reservoirs and from geogenic impacts on water chemistry for pit lakes in coal and ore mining districts. The state of reservoir building and the problems are presented by the World Commission on Dams (<http://www.dams.org>). Reviews and other comprehensive scientific publications on mining lakes are in an initial phase (50).

REGIONAL PROGRAMS FOR PROTECTING LAKES AND COLLECTING DATA

The protection of lakes and other surface waters is different regionally. The Environmental Protection Agency (EPA) has set standards in North America, and the

Table 2. Basic Data on Ancient Lakes^a

	Surface Area, km ²	Maximum Depth m	Water Volume, km ³	Age & Origin, Years
<i>African Rift Valley</i>				
Tanganyika	32,900	1471	18,900	20 mio, tectonic
Malawi	22,490	709	6,140	2 mio, tectonic
Victoria	68,460	92	2,700	20,000, tectonic
<i>Asia</i>				
Dead Sea	1,020	433	188	tectonic
Caspian	374,000	1025	78,200	>5 mio, tectonic
Baikal	31,500	1637	23,000	20 mio, tectonic
Aral	64,100		1,020	>5 mio, tectonic
Issyk-Kul	6,240	702	1,730	25 mio, tectonic
Biwa	681	104	27.6	2 mio, tectonic
<i>N. America</i>				
Tahoe	500	501	156	2 mio, tectonic
<i>S. America</i>				
Maracaibo	13,010		280	36 mio, tectonic
Titicaca	8,562	284	903	2.8 mio, tectonic

^a References (8,44–49).

European Union in the EU countries. The regulations of the EU-Water Framework Directive have initiated many activities to register, describe, and improve surface waters and their state in Europe (see documents of the European Environmental Agency (EEA) and EU-Commission). There are data collections on world lakes for some regions and for selected countries: ILEC, EEA and EU-documents, EPA-documents (8,12,13,15,16,21,42,51–71) (31,32,72–75). For a series of important single lakes, there are monographic descriptions (11,76–80).

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THE THEORY OF ALTERNATIVE STABLE STATES IN SHALLOW LAKE ECOSYSTEMS

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OVERVIEW

The idea of alternative stable states was first proposed in the late 1960s (1), and within ecological communities, it was described mathematically in the 1970s (2,3). The contemporary view of shallow lakes is that two alternative stable states can exist over a range of nutrient concentrations: a clear water state dominated by submerged macrophytes and a turbid phytoplankton-dominated state (4–8). Unique states may be found at either end of the nutrient continuum, but over a wide range of nutrient concentrations (<50 to several thousand $\mu\text{g l}^{-1}$ TP), either of these two states can exist, stabilized by a number of buffer mechanisms (5,6,9–11).

A “ball and cup” diagram is a simple way to visualize this idea (Fig. 1). The position of the ecological system (the ball) can change between the two states and settles at the bottom of the cup (stable equilibrium). The slope of the cup determines both the speed and the direction of the system change to a stable state. At either low or high

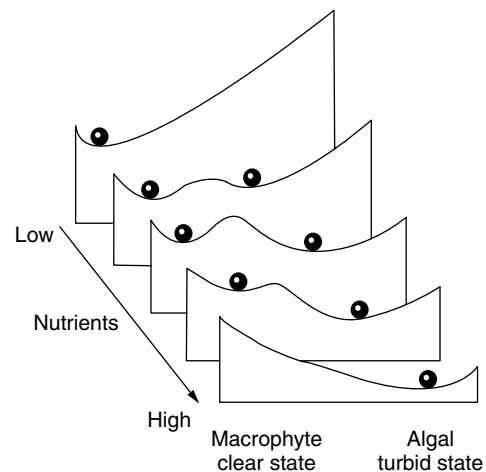


Figure 1. “Ball and cup” schematic of the potential for two stable states (clear water macrophyte-dominated versus turbid algal-dominated) to exist within a shallow lake under a wide range of nutrient conditions.

nutrient levels, one of the two stable states exists—the ball can only be in one location because of the slope of the horizon. However, between these points the system can rest in either cup (state). The ease with which one stable state can be switched to the other is indicated by the position of the two balls with respect to one another and the slopes that separate them. Therefore, if the starting position is a pristine, low-nutrient (oligotrophic), macrophyte-dominated, clear water lake, the nutrient levels can increase substantially before the only possible scenario is a turbid algal-dominated lake. However, to reverse the situation takes more than just reducing the nutrients as the ball can remain in the turbid state even at relatively low nutrient levels. To switch the state prematurely requires that a force be applied to the ball to push it over the edge of the algal cup and into the macrophyte state.

Therefore, each stable state has a number of buffer systems that help maintain that state, and through top-down and bottom-up processes, fish, plankton, invertebrates, macrophytes, and nutrients are all entwined in reciprocal feedback mechanisms that not only determine the state that is present (12,13) but can also be manipulated to force a shift.

Macrophyte buffer mechanisms that help maintain and stabilize a clear water state in shallow lake systems are numerous (4,14–16). Macrophytes act as habitats and refugia for macroinvertebrates and cladocerans that reduce the epiphyton and phytoplankton communities by grazing (6,17). Davis (18) suggested that the plant bed environment, often deoxygenated, favors grazers by discouraging their predators. Macrophytes may also release allelopathic chemicals (19); partake in the “luxury uptake” of nutrients, removing available nutrients for phytoplankton (20–22); and reduce available nitrate by anaerobic decomposition processes such as denitrification (23). Moreover, certain species of macrophytes have been shown to oxidize the sediment and reduce the release of phosphorus. With less available phosphorus in the water column, phytoplankton populations are reduced (24,25). In addition, stands of macrophytes reduce water movement within them. In consequence, suspended sediment resettles and turbidity falls, aiding the growth of macrophytes (26). Moss (10) suggests that even if light penetration is falling, submerged macrophytes may switch from low growing forms to taller species and that plants heavily encrusted with epiphyton may be able to shed their leaves and produce new growth. These processes competitively disadvantage phytoplankton and can buffer the macrophyte-dominated state.

Once established, on the other hand, phytoplankton can buffer the switch back to a macrophyte state by growing much earlier in the season than macrophytes in temperate regions. In doing so, they can curtail the development of turions, rhizomes, or seeds by shading in spring or reduce the formation of propagules by shading and competition for CO₂ in late summer (10). Submerged plants require carbon for growth, but algae have shorter CO₂ and HCO₃ diffusion pathways owing to their small size, thus removing the carbon available to the bulkier macrophytes (27). Some researchers have suggested that certain blue-green algae

may even release chemicals toxic to macrophytes (28). In addition, phytoplankton-dominated open water has few refugia for grazing Cladocera; thus, any Cladocera venturing forth are typically removed by zooplanktivorous fish, which may be compounded by the fact that, with reduced macrophytes, few large macroinvertebrates would be available to large fish, favoring smaller sized fish feeding largely on zooplankton (6). Finally, in poorly flushed systems, phytoplankton species can consist of large filamentous or colonial blue-green algal species that are inedible to zooplankton, which along with the above mechanisms, all help to maintain phytoplankton-dominated water states over a wide range of nutrient concentrations.

By understanding the mechanisms that buffer and thus maintain the alternate stable states, it is possible to attempt to force a shift or switch from one state to another by manipulating top-down and bottom-up processes. Such attempts are often made during lake restoration projects when a desire exists to shift an algal-dominated turbid lake into a clear water macrophyte-dominated state. However, the long-term success of lake restoration projects, although they use knowledge of top-down and bottom-up processes, is not guaranteed and unexpected results are common (29–31). The uncertainty that surrounds these outcomes often stems from a lack of understanding about the buffer systems that stabilize the alternate states within the specific lake under restoration and an oversimplification of the top-down and bottom-up processes involved (13).

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NOAA LAKE LEVEL FORECAST FOR LAKE MICHIGAN RIGHT ON TARGET

NOAA Great Lakes
Environmental Research Lab



July 17, 2003—How high or low the water level is in the five Great Lakes has long been of interest to those who use the lakes for commercial and recreational boating and fishing, energy, and as a source of drinking water. The levels have been measured and forecast monthly for many years, and NOAA scientists hit the bull's-eye with their July forecast of the water level of Lake Michigan.

"We were within inches of the actual measurement," said Cynthia Sellinger, deputy director of the NOAA Great Lakes Environmental Research Laboratory in Ann Arbor, Mich.

GLERL uses meteorological forecasts from the NOAA National Weather Service to forecast the lake levels six months out. Back in January, GLERL's Lake Michigan forecast was 176.61 meters—about 579 feet; in July the actual level was 176.65 meters—approximately 2 inches higher than the forecast. Sellinger meets monthly with her



colleagues from Environment Canada and the U.S. Army Corps of Engineers to forecast the levels.

Since 1997, Lake Michigan has dropped 4.1 feet, the largest drop in that short of time span since records have been kept. Lakes Michigan and Huron are often considered one body of water as they are connected by the Straits of Mackinac, located between the upper and lower peninsulas of the state of Michigan.

The other lakes are more difficult to forecast, because the water flow of Superior and Ontario are regulated by power companies and Erie gets 80 percent of its water from Michigan and Huron, Sellinger explained.

The Great Lakes—Superior, Michigan, Huron, Ontario, and Erie—are the major bodies of water within the Great Lakes Basin. The basin is located along the international boundary between Canada and includes portions of eight states and the provinces of Ontario and Quebec. Combined, these lakes make up the largest surface fresh water body in the world—they contain 90 percent of the U.S.'s fresh water and 20 percent of the world's fresh water.

The Great Lakes carry thousands of recreational and commercial ships annually. Lake levels are important to shippers, who must load the lake carriers with fewer goods when the levels are low and to recreational boaters, who often have to find dock space in deeper water. The Lakes provide drinking water to the 40 million residents as well as water to power and cool electric generating plants.

Lake levels have been dropping for the past six years and lake evaporation has increased significantly.

"The only good thing that comes out of lower lake levels is more beach for beach-goers," according to Sellinger. However, that also exposes and destroys some vital breeding habitat for lake wildlife.

Lake levels have been forecast before GLERL was formed in 1974. Part of the laboratory's mandate is to provide the lake level forecasts, a task that was given to GLERL when the research arm of the U.S. Army Corps of Engineers was transferred to the laboratory.

NOAA is dedicated to enhancing economic security and national safety through the prediction and research of weather and climate-related events and providing environmental stewardship of the nation's coastal and marine resources. NOAA is part of the U.S. Department of Commerce.

RELEVANT WEB SITES

NOAA Great Lakes Environmental Research Lab

NOAA Great Lakes Water Levels
NOAA Great Lakes Satellite Information

MEDIA CONTACT

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(Photo courtesy Greg Lang of the NOAA Great Lakes Environmental Research Laboratory.)

SUBMERGED AQUATIC PLANTS AFFECT WATER QUALITY IN LAKES

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BACKGROUND INFORMATION

A principal cause of water quality impairment in lakes and reservoirs is enrichment by excessive levels of nutrients, resulting in undesirable conditions that include noxious and/or toxic algae blooms (1), and taste and odor problems (2). The nutrient content of lake water is determined by a myriad of factors, including external loading rate, water residence time and depth, geochemical interactions in the water body, and biological processes. Biological processes are particularly important in shallow lakes. Fish and benthic animals recycle nutrients from bottom sediments, buoyant algae carry nutrients upward in the water column, settling rates of algae determine loss rates for water column nutrients, and plants facilitate various nutrient-scavenging processes (3,4). Submerged plants, in particular, can have major influences on water quality in terms of clarity, biomass of noxious algae, taste and odor, organics, and suspended solids—factors of major relevance to society's use of the water.

This chapter addresses four questions:

1. What types of submerged plants occur in lakes, and how does type of plant relate to water quality benefits?
2. What factors control the biomass and spatial extent of submerged plants?
3. What are the processes whereby plants affect lake water quality, in particular, the concentrations of nutrients, turbidity, and the occurrence of noxious algae?
4. How can lakes be managed to promote the development of submerged plants that will help to improve water quality?

For a detailed overview of the ecological role of submerged plants in shallow lakes, the reader may consult Jeppesen et al. (5) or Scheffer (3). For identification of different types of plants in lakes and for general information about their biology and distribution, there are a number of web-based resources provided by major universities; for example: <http://aquat1.ifas.ufl.edu> (University of Florida, Center for Aquatic and Invasive Plants).

TYPES OF SUBMERGED AQUATIC VEGETATION RELATIVE TO WATER QUALITY BENEFITS

Lakes can support rooted vascular plants (true plants that have tissues for gas and materials transport); plant-like macroalgae, such as *Chara* (shrimp grass) that do not have vascular tissues or roots; and filamentous algae that can form dense mats on the lake bottom, sometimes covering rocks and even other plants. Filamentous algae are least desirable from a water quality standpoint, because at high biomass, they can kill other plants by shading and they form floating noxious mats at the lakeshore when they decompose (3). *Chara* generally is a pioneer species that rapidly invades lake habitats after a disturbance (6,7), and because it does not have roots, all of its accumulated nutrient content may be recycled back into the water column if unfavorable conditions (e.g., reduced light, high wind, or cold water) cause the plants to be lost. Certain vascular plants, such as *Ceratophyllum* (coontail), also do not have roots and would have this same problem. In regard to *Chara*, note that clear water and low nutrient levels are most often associated with dense beds of this plant in temperate (8,9) and subtropical (7) lakes and that processes other than direct incorporation of nutrients into biomass may be most important for water quality improvement by *Chara* beds. Rooted vascular plants—common examples in North America are *Vallisneria* (eelgrass) and *Potamogeton* (pondweed)—are more permanent than *Chara*, and because they can transfer nutrients to their underground roots prior to seasonal “die-off,” they have potential to provide a better long-term nutrient sink.

FACTORS CONTROLLING THE BIOMASS AND EXTENT OF SUBMERGED VEGETATION

Scientists and lake managers have given considerable attention to the factors that control the biomass and spatial extent of submerged plants in lakes. Obviously, water depth plays a key role because ultimately it determines the amount of light that reaches the lake bottom where plants germinate from seeds (or oospores, in the case of algae) and begin to grow. However, light availability at any given depth is heavily influenced by the turbidity of the water, which is determined by the amount of suspended solids, algal cells, and dissolved organic matter. Hence, a number of authors have identified significant relationships between the maximal depth for submerged plant occurrence in lakes and simple indicators of turbidity, such as Secchi disk transparency (10–13). The relationships vary from lake to lake. For example, in Lake Okeechobee, Florida, the published models predict that submerged plants can occur at depths between 0.7 and 1.8 m, whereas plants actually occur at depths as great as 2.0 m. This may reflect the occurrence of plants that are adapted to low light levels in this turbid shallow lake (14). In general, *Chara* can occur at greater depths than vascular plants, for any given level of turbidity (15).

Experimental studies have confirmed the causal effect of underwater light availability on submerged plant growth (14,16). However, when evaluating field data, it

can be difficult to tell whether plants are responding to variations in underwater irradiance or whether they are the cause of high irradiance in certain areas of a lake because they have various functions that reduce turbidity. It is a classic “chicken versus egg” paradox. Some shallow lakes display zones of clear water immediately adjacent to turbid water of the same depth (8), the only difference is that the clear areas have dense beds of submerged aquatic vegetation (SAV). In such instances, one might conclude that the plants have caused the clear water. In Lake Okeechobee, Florida, clear water expanded over an ever-increasing area during summer 2000, coincident with the expansion of large beds of *Chara*. Areas adjacent to the plant beds had more turbid water, but they also had slightly greater depths (6). It was not possible to establish a cause–effect relationship. Scheffer (3) provides a good overview of the issue of correlation versus causation as it pertains to SAV and turbidity in lakes, for those interested in more information on this subject.

In addition to depth and turbidity, sediment type may influence the distribution and biomass of submerged plants. Soft sediments that are readily suspended by wind, waves, and fish may not support plants, especially types like *Chara* that do not have roots. In Lake Okeechobee, which has areas of sand, peat, and mud sediments, *Chara* does not occur at locations that have soft organic mud, even when depth and light conditions appear to be favorable (7). A simple empirical model, based on water depth and sediment type, was able to predict at 75% accuracy the spatial distribution of *Chara* over a 170 km² landscape in that lake. The sediment type found also was the major factor controlling where plants could occur in Lake Kinneret, Israel (17), whereas water depth was the factor controlling the timing of plant occurrence.

Other factors that can influence the distribution and biomass of submerged plants include the degree of exposure to wind and waves (13,18), the slope of the lake bottom (19), and as mentioned before, overgrowth by filamentous algae (20).

PROCESSES WHEREBY SUBMERGED VEGETATION AFFECTS WATER QUALITY

When one compares lakes with and without dense submerged plants or follows the conditions in a lake over time as plants increase, striking changes in water quality are apparent. Water can change from murky green (algae blooms) to crystal clear (Fig. 1), and at the same time, nutrient levels in the water can dramatically decline. Even a relatively low biomass of submerged plants can attenuate peak concentrations of suspended algae (phytoplankton) in a lake’s water column (21; Fig. 2). The benefits of clear water and plants for the fish, wildlife, and humans who use the resource are obvious.

Submerged plants mediate changes in water quality by a variety of mechanisms. The focus here is on processes that control nutrient and light availability, rather than food web interactions. However, it should be recognized that a potentially important function of submerged plants is to provide refuge for large zooplankton (22). During the daytime, large zooplankton can escape fish predation by

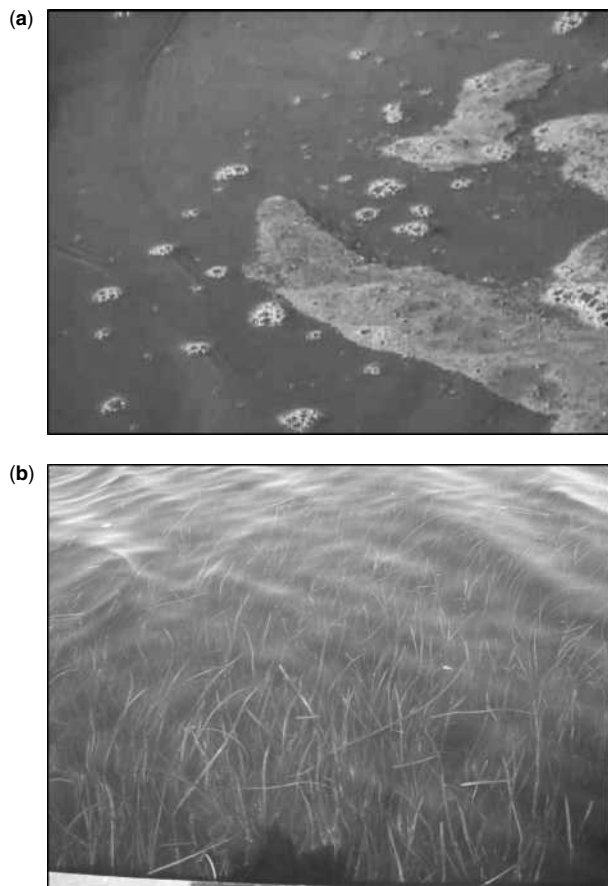


Figure 1. Photographs of typical conditions in turbid, phytoplankton-dominated lakes that have high levels of nutrients in the water column and blooms of algae (a), versus clear, plant-dominated lakes that have low nutrient levels in the water and no algal blooms. These photographs were taken by the author at Lake Okeechobee, Florida.

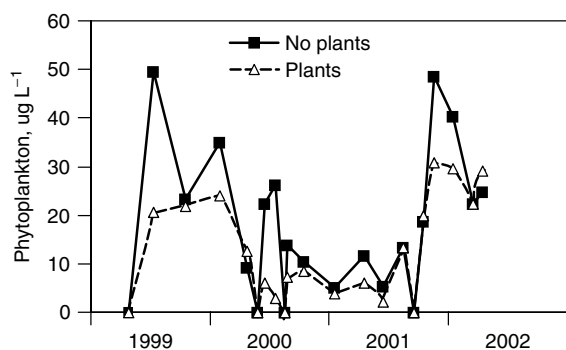


Figure 2. Data from the shoreline region of Lake Okeechobee, Florida, showing the concentrations of phytoplankton chlorophyll ($\mu\text{g L}^{-1}$) at sampling stations with plants versus stations without plants. The presence of plants dampens the peaks in phytoplankton, as a result of various processes described in the text. Modified from a figure in Havens (20).

migrating into the plant zone, and at night, when they are not susceptible to those visual predators, they can migrate out into the open water to graze on phytoplankton. The result is theoretically a reduction in algal biomass

and reduced risk of nuisance algal blooms, although uncertainty remains about the magnitude of this effect. One of the complicating issues is that plants may also provide refuge for the small fish that eat zooplankton (22).

In a conceptual framework (Fig. 3), one mechanism whereby submerged plants can benefit water quality is the uptake (U) of dissolved nutrients that otherwise could fuel algal blooms and the array of adverse impacts shown in the diagram. Benthic algal mats can also take up nutrients from the water, further reducing availability for the bloom-forming phytoplankton. Much of the nutrient uptake by plants actually is associated with attached algae (periphyton) that grow on plant surfaces (23,24). Vascular plants and benthic algae also take up nutrients from lake sediments; it has been documented that benthic algal mats can substantially reduce the diffusive flux (D) of nutrients from sediments to the overlying water in both shallow freshwater and estuarine systems (25–27). All of the plants and algae mentioned here can at times excrete (E) dissolved nutrients back into the sediment or water column. Conditions that favor net growth of plants and/or benthic algal mats favor a net uptake of nutrients, which translates into better water quality.

Nutrients present in the water column or sediments in a dissolved form can also be transformed (T) into particulates that ultimately settle (S) to the lake bottom at a rate that depends on water column mixing (or lack thereof), the density of the water, and the characteristics of the particles. Plants can stimulate both the transformation of dissolved nutrients to particulates in the water and the rate of settling. In regard to transformation (T), when plants carry out intense photosynthesis, the pH of the surrounding water can be substantially elevated. At high pH (e.g., >10), much of the dissolved inorganic carbon (C) occurs as carbonate ions $(\text{CO}_3)^{2-}$ (28), which can be precipitated to the sediments, or onto the surface of plants, as calcium carbonate (CaCO_3). More important, from a water quality standpoint, is that this process can be accompanied by coprecipitation of phosphorus (29), as insoluble hydroxyapatite $[\text{Ca}_{10}(\text{PO}_4)_6(\text{OH})_2]$. Phosphorus is a nutrient that often is limiting to phytoplankton in freshwater lakes (30). Therefore, coprecipitation could help reduce phytoplankton growth and suppress formation of blooms. However, it must be noted that at high pH, the release of dissolved phosphorus from lake sediments (D) is sometimes enhanced, even under oxidized conditions (31). Depending on circumstances, this could counteract the effects on particle transformation.

Plants play an important role in settling of particles (S) by slowing the rate of water movement (32), and acting as sediment “traps” by intercepting particles using their leaves and stems; the particles subsequently sink to the lake bottom (33). The uptake of nutrients by periphyton, which ultimately provide particulate material (dead cells) to the sediments is another loss process mediated by plants. Once the particles settle within a plant bed, they are less likely to be re-suspended (R) because dense beds of submerged plants significantly reduce the amount of wind and wave energy that reaches the sediment surface (34).

Finally, it has been suggested that submerged plants might inhibit the growth of suspended algae by releasing

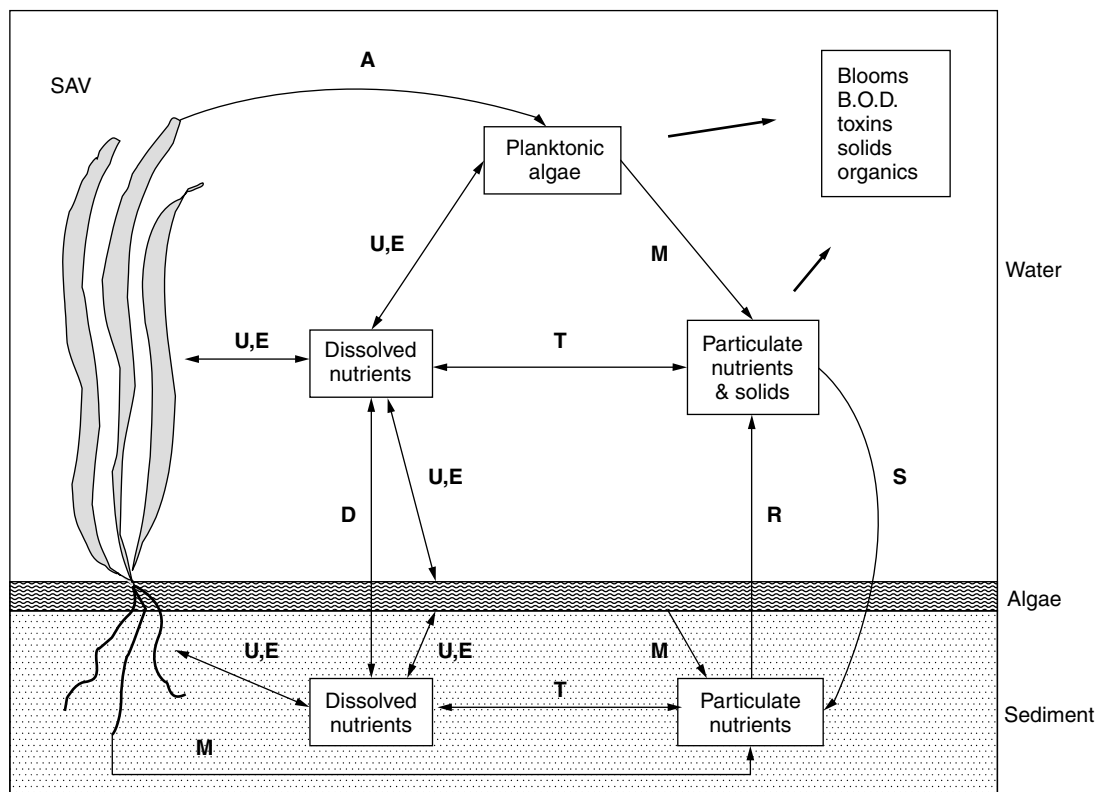


Figure 3. A simple conceptual model, illustrating how submerged aquatic plants and benthic algal mats influence the dynamics of phosphorus in a typical eutrophic lake. The arrows correspond to various processes: **A** = release of allelopathic chemicals; **U** = uptake of dissolved nutrients; **E** = excretion of dissolved nutrients; **T** = transformation of nutrients from dissolved to particulate form, or vice versa; **D** = diffusion of dissolved nutrients from sediments to water, or vice versa; **M** = mortality; **S** = settling of particles; and **R** = resuspension of particles by wind, waves, or bioturbation. B.O.D. = Biological Oxygen Demand. The two large arrows point to impacts of algal blooms and solids on the water resource. This figure is modified from a similar model presented in Havens et al. (23).

allelopathic chemicals (**A**) into the water column (35). This effect has been documented in controlled tank studies with a high biomass of plants and restricted water volume, but uncertainty remains regarding its importance in lake ecosystems.

MANAGING LAKES TO PROMOTE SUBMERGED VEGETATION

The total biomass of primary producers in a lake is controlled by nutrient input to the ecosystem (36), whereas partitioning of that biomass into phytoplankton and plants is determined by a complex set of factors. Understanding how particular environmental conditions influence submerged plants is critical to facilitating plant survival and growth, and in turn, maintaining good water quality (see below). Lakes dominated by soft organic sediments or heavily influenced by wind-driven waves are not well suited for submerged plants. However, even under these conditions, options may exist, such as dredging to remove soft mud or construction of wave barriers to provide protected areas for plants to grow (37). In lakes that have favorable sediments and low to moderate wave energy along the shoreline, the key factor controlling plants is

likely to be light availability, which as noted before, varies with the depth and turbidity of the water. One of the factors controlling turbidity is the nutrient content of the water, which determines the abundance of phytoplankton cells that absorb light. Scheffer (38) noted that there is a complex relationship between nutrients and turbidity in lakes that varies with the presence or absence of plants. This relationship (Fig. 4) is highly relevant when considering how to establish plants in a nutrient-rich, turbid lake.

As a lake becomes enriched with nutrients, the turbidity of the water increases. In lakes with plants, the turbidity at any given nutrient level is lower than that in lakes without plants due to the mechanisms described before (Fig.3). Once a "critical turbidity" is reached, where light does not allow net growth to occur, plants are lost, and turbidity quickly increases (upward arrow), as the nutrients formerly sequestered by plants go into the biomass of phytoplankton in the water. To restore plants to a lake that has reached this state, nutrients must be reduced to a lower level than where the switch previously occurred (downward arrow). Both the clear-water, plant-dominated state and the turbid-water, phytoplankton-dominated state are highly stable—it takes a large

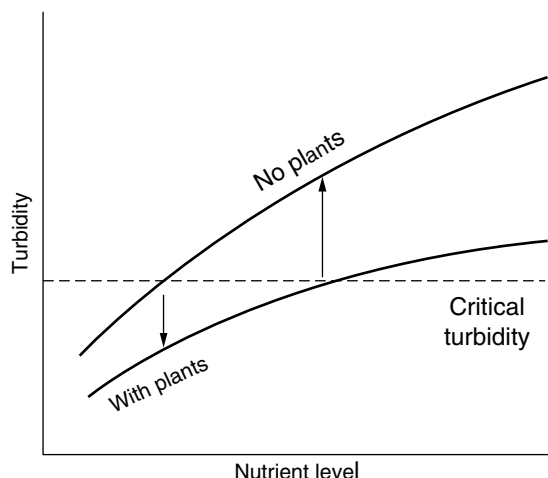


Figure 4. Relationship between turbidity and nutrient level in lake water, in lakes with versus without plants. Lakes with plants have a lower turbidity for any given nutrient level than lakes without plants. When nutrient levels increase in lakes with plants, the plants can be lost once a critical turbidity is reached, at which point the net growth of plants is zero. When this occurs, turbidity can rapidly increase (upward arrow). To restore the plants in such a lake (downward arrow), the nutrient level must be substantially reduced. This figure is modified from a diagram in Scheffer (3).

input of nutrients to switch a plant-dominated lake to a phytoplankton lake and an even larger reduction of nutrients to switch the lake back (3,38). This stability of states reflects the biological feedback loops that occur in the system. Plants cause clear water, which favors more plants, which cause even clearer water...etc. Phytoplankton cause turbid water, which reduces plants, which favors more phytoplankton, which further increases turbidity and so on.

Note that at extremes along the nutrient axis (extremely low or high nutrient levels), plants can exist in only one state—clear or turbid, respectively. However, for lakes that have moderate to high levels of nutrients and the possibility of alternative stable states (38), one method for switching a lake from phytoplankton to plant dominance is to reduce inputs of limiting nutrients, such as phosphorus and nitrogen (3,39). Ultimately, this is the path to lake restoration. However, an interim management strategy that can sometimes be used in tandem with nutrient reduction is to lower the lake's water level. Critical turbidity is a function of depth, so that in a deep lake, relatively low turbidity might be sufficient to prevent plant growth. However, in shallow water, enough light may reach the lake bottom to allow plants to germinate and grow. By lowering the water level, it is possible to raise the critical turbidity (Fig. 5), so that plants can become reestablished at a higher level of nutrients than that when the lake had deeper water. At this point, the lake has essentially been reset onto the lower curve.

A switch from turbid to clear water happened in the shoreline region of Lake Okeechobee, Florida, in summer 2000, after an intentional lowering of the water level was followed by a major drought (40). The water level in the lake dropped by more than 2 m, and the

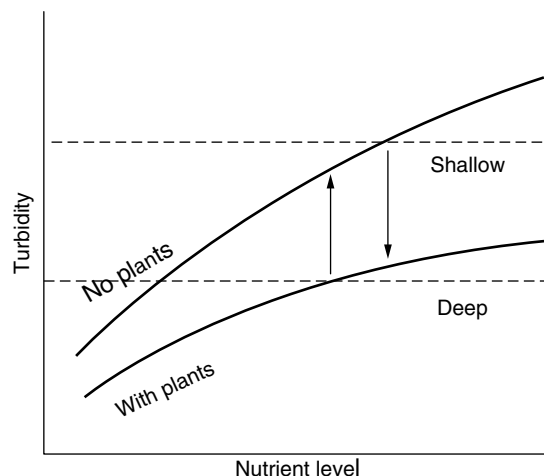


Figure 5. Effects of water depth on the occurrence of plants in turbid lakes. By substantially lowering the water level of a lake, the critical turbidity is increased (upper dashed line), and this can allow plants to recover (downward arrow), even though the nutrient level remains high. If the water level increases substantially (lower dashed line), the plants may be lost, even at a lower level of nutrients than that where they were recovered (upward arrow). Unless there is capability for strong control of water levels and no other adverse impacts of keeping the lake low (e.g., impacts on water supply), the ultimate solution to keeping plants and clear water in a lake is to reduce external nutrient inputs. This figure is based on information in Scheffer (3).

spatial extent of submerged plants increased from less than 10 to over 170 km² (16). Water in shoreline areas, which was highly turbid from algae blooms in the late 1990s, became crystal clear in 2000–2002. Recently, it has somewhat deteriorated, as higher water levels led to reduced biomass of plants. Ultimately, reduced nutrient input is the solution to the water quality problem in this and other nutrient-enriched lakes (39), because as long as alternative states exist, uncontrollable events, such as marked increases in water level or wind storms, could potentially shift a plant-dominated lake back to a turbid lake with phytoplankton blooms.

SUMMARY

Water quality in shallow lakes and reservoirs can be strongly influenced by biological processes, including those associated with fish, invertebrates, algae, and plants. The most striking influences often are those associated with submerged aquatic plants, which include rooted vascular plants, loosely attached macroalgae, such as *Chara*, and filamentous benthic algal mats. Where conditions favor high biomass and spatial extent of these plants, water turbidity generally is low, biomass of phytoplankton is low, and quality of water for drinking and other uses is good. The plants maintain these conditions by removing nutrients from the water; supporting attached algae that also sequester nutrients; stabilizing bottom sediments; and a variety of other physical, chemical, and biological processes. Actions to favor development of submerged plants in shallow, nutrient-enriched lakes can help to

maintain good water quality, although the long-term solution still is to reduce external nutrient inputs.

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LAKES—DISCHARGES TO

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Lakes have suffered from a long history of receiving pollutants from various sources. These pollutants degrade water quality, with implications for aquatic and terrestrial ecosystems as well as for a variety of issues for human populations dependent on that water and those ecosystems. The degree to which these pollutants degrade the lake depends on the amount and types of pollutants as well as the physical characteristics of the lake. For an introductory overview of how discharges to lakes are harmful, the pollutant sources and how pollutants are transported into the lake are described. The physical characteristics that determine the assimilative capacity of a lake are then covered. Finally, the effects that different pollutants have on lakes and their dependent ecosystems are explained as well as the ramifications for aquatic species, birds, wildlife, and humans.

POLLUTANT SOURCES AND TRANSPORT

Polluted discharges into lakes can come from watercraft, terrestrial sources, and atmospheric deposition. Watercraft are categorized into ships, boats, and recreational equipment. Terrestrial pollutant sources can be categorized into point and nonpoint sources. Point sources of pollution emanate from a single location, like a factory's pipe dumping effluent into a stream or lake. Nonpoint sources of pollution, often called "polluted runoff," can come from all over a landscape and from many locations at once, such as fertilizer runoff from suburban lawns or oil washing off city streets. Atmospheric deposition has characteristics of both point and nonpoint source pollution and so is treated separately here. Atmospheric deposition of pollutants originating from smokestacks travels through the air and then falls into the water.

Watercraft

Ships discharge a variety of pollutants into lakes. The ballast water that ships discharge into the water contains

oils and invasive species. The entry of the zebra mussel into the Great Lakes is a good example of how an aquatic ecosystem can be significantly changed by the introduction of just one new species. Now, regulations require ships to dump their ballast water in the open ocean before entering the Great Lakes. Used oil from ship engines has also been dumped into lakes for years. Sewage from ships has also been dumped untreated into lakes, although cruise ships have recently been pressured to install onboard sewage treatment systems so that only treated effluent is dumped. Atmospheric deposition from smokestacks is becoming a more serious problem with increased global shipping. Ships burn low-grade diesel fuel and these carcinogenic compounds then enter lakes.

Boats used for recreation also pollute lakes. Boat owners may dump their sewage straight into the lake rather than pumping it out dockside at sewage receptacle facilities. Gas spills often happen at docks when gas pumping is not done with care, and some gas and oil are emitted into the water from outboard motors.

Smaller recreational equipment like jet skis pollute water with oil and gasoline at a far higher rate than outboard motors. Jet skis also damage littoral habitat, kill aquatic animals, or frighten aquatic species and disturb mating or nesting behaviors. Chromosomal damage in fish has been linked to the fossil fuel discharges from jet skis.

Terrestrial Point Sources

The most common point sources polluting lakes are sewage systems and industrial effluent pipes. Sewage can enter lakes either through intentional discharge of raw or treated sewage, or through occasional discharges from combined sewer overflows (CSOs). Depending on the city and on the country, one can find cities discharging raw sewage into lakes rather than spending the money to treat it properly. Sewage treatment plants in other cities may have primary, secondary, or tertiary sewage treatment, resulting in various types and concentrations of pollutant discharge into lakes or their tributaries. Combined sewer overflows are found in cities where stormwater and sewage flow down the same pipes, rather than flowing through separate stormwater and sanitary sewers. In rain events, the additional stormwater entering sewage pipes may be more than a sewage treatment plant's designed capacity. When this happens, sewer pipe outlets are designed to open and discharge raw sewage into a local stream or lake to prevent too much sewage from reaching the treatment plant; passing through the plant without sufficient treatment; and/or backing up into the city. Consequently, the raw or partially treated sewage from some cities contributes a daily material loading to a lake, whereas combined sewer overflows discharge sporadically as a result of heavy rain or rapid snowmelt. Some of the constituents of sewage are human waste, bacteria, viruses, chemicals, detergents, heavy metals, and pharmaceutical and personal care products (PPCPs) (1) such as antibiotics, steroids, birth control pills, shampoo, hair dye, and perfume.

Industrial pollutants enter a tributary or lake by a pipe, either intentionally through a permitting program or by accidental release. In the United States, any

industrial facility, including sewage treatment plants, must have a National Pollutant Discharge Elimination System (NPDES) permit to discharge anything into a water body. The type, quantity, and concentration of pollutants from various industrial processes are written into the permit, thus limiting the effluent discharge to the permit limits. Pollutants may be chemical, biological (e.g., bacteria, viruses, or organic matter that deoxygenates the water), thermal, radioactive, metals, and so on. Because there is a permitting system, and since the name implies a reduction or elimination of pollutant discharges, it is widely assumed that permits issued are still within the assimilative capacity of the water body, and that the appropriate government agencies monitor and enforce permits to ensure that the water is not becoming degraded. Unfortunately, this is not the case, and rivers and lakes are receiving rising material loads.

Terrestrial Nonpoint Sources

Nonpoint source pollutants move into a lake from its entire drainage area. Nonpoint sources are grouped into urban runoff, mining, agriculture, silviculture, groundwater, and human transport. Urban runoff includes fertilizers, pesticides, and pet waste from lawns; oil, rubber, deicing chemicals, and metals from roads; and sediment from construction.

Mining contributes acid mine drainage to tributaries or lakes or can put cyanide into the water when it is used to leach gold out of gold ore. Strip mining transports sediment into streams. Agriculture contributes a significant portion of the overall nonpoint source pollutant load to lakes. These pollutants include sediment from erosion, pesticides mobilized from the soil by rain, and overapplication of commercial fertilizer or manure.

Silviculture primarily contributes pesticides and sediment to lakes. Pesticides used on tree farms or areas sprayed for infestations (e.g., gypsy moth) are transported through rain and snowmelt into water bodies. Sediment from road building for timber trucks or from clear-cutting or harvesting on steep slopes is also discharged into streams and lakes.

There are two primary types of pollutants moving through groundwater. Leaking septic systems contribute organic sewage material, *Escherichia coli*, viruses, and toxic household cleaners. Acid rain filters into soils, and the resulting chemical reaction then mobilizes mercury, manganese, aluminum, and zinc into the groundwater and then into lakes through subsurface flow. Humans can transport invasive species unintentionally or with purpose. For example, zebra mussels can “hitchhike” on a pleasure boat being hauled from one water body to another, or canals have been built that connect previously distinct ecosystems and allow migration of species into a new lake. Intentional transfer of invasive species happens when people get rid of pet fish by flushing them down the toilet or dumping them directly into streams or lakes, and this fish will now be an exotic species for that lake. Other exotic species can be placed in a lake with the intent of using them to control other aquatic animals or plants, but several of these experiments (e.g., carp) have

backfired; new unexpected problems arise from the exotic introduction.

Atmospheric Deposition

Although some scientists may regard atmospheric deposition as a type of nonpoint source pollution, it is categorized here as distinct from terrestrial point or nonpoint source pollution because it comes from discrete sources (smokestacks), but from numerous locations. The type of airborne pollutant discharged from smokestacks depends on the type of industrial facility, the type of energy they use, and their chemical processes. For example, electrical generation plants burning coal, or other industries that use coal for their energy source, can emit lead, mercury, zinc, arsenic, uranium, sulfur dioxide, and nitrogen oxides. All of these can enter the water surface either through wet (rain or snow) or dry deposition. The heavy metals, radioactive uranium, and acid rain constituents have different effects on lakes. Downwind lakes close to these smokestacks will receive a heavier loading of heavy metals, whereas the SO_2 and NO_x that contribute to acid rain can travel much farther and degrade more distant lakes. Other industries emitting other kinds of chemicals or metals into the air will affect lakes according to how those metals or chemicals interact with lake chemistry or lake biota.

Atmospheric deposition of mercury is bioconverted by bacteria into methylmercury, a fat-soluble neurotoxin. It then moves up through and beyond the aquatic food chain. Regions receiving deposition of both mercury vapor and acid rain will result in an even higher conversion of elemental mercury in methylmercury, moving first into the water and then into the fish, wiping out entire fish species. Acid rain also mobilizes aluminum, manganese, and zinc from soils into the lake.

PHYSICAL CHARACTERISTICS OF LAKES

What is the assimilative capacity for a lake for certain types or amounts of pollutants? A lake's ability to absorb pollutants without degrading its biological and chemical processes is a function of its physical characteristics. Some limnological characteristics are described here. (For more information into lake processes and response to pollutants, see Reference 2.) What is the lake's volume and depth? The greater the lake volume, the more it can dilute pollutants down to less harmful concentrations. Vertical temperature differences found in deep lakes can stratify water into layers. This stratification, and whether or not the lake “turns over” (a mixing of the layers by turbulence in the water) in spring and fall, controls the amount of oxygen, nutrients, and other chemical cycling occurring in the upper or lower waters of the lake.

Therefore, stratification influences which types of pollutants may stay or have an effect only in the upper strata, only in the lower strata, or whether pollutants have access and impact throughout the entire water column and the sediment below. Because stratification controls temperature, it controls the rates of biochemical reactions, and these rates may also provide a buffering effect against

some pollutants. Similarly, because stratification controls oxygen levels, pollutant loadings of organic materials (e.g., paper pulp, sewage effluent, manure) will have more negative impact during the times of year that oxygen levels are lowest. Consequently, the same daily amounts of material loadings to a lake can have different effects on the lake depending on the season.

The flushing rate of a lake describes how long it takes for new water inputs to the lake to entirely replace existing lake water with new water. How often the lake theoretically refills with new water affects how fast some of the pollutants entering a lake may be flushed out before they can harm lake processes very much. The higher the flushing rate, the more diluted some pollutants can be because some pollutants will not be able to stay in the lake for long.

The typology of a lake, whether it is naturally more oligotrophic or eutrophic, affects its ability to absorb some pollutants without changing lake characteristics significantly. Over hundreds or thousands of years, a lake naturally receives more nutrient inputs and naturally undergoes eutrophication, or nutrient enrichment, and slowly changes from oligotrophic to eutrophic. When rapid nutrient inputs are due to human action, this is called cultural eutrophication. A more oligotrophic lake is typically deep and has very clear green or blue water, high oxygen, few nutrients or calcium, and limited phytoplankton or other plant species. At the other end of the spectrum is a eutrophic lake, which is shallow and has opaque green or brownish green water, very low oxygen in summer, high nutrients and calcium, and frequent algal blooms in reaction to the high nutrient concentration. So, for example, a naturally oligotrophic lake would become more eutrophic in response to high sewage effluent or farm manure inputs, and a more eutrophic lake would have less assimilative capacity for high loadings of sewage effluent, since the lake is already heavily loaded with nutrients.

POLLUTANT FATE AND CONSEQUENCES

The various sources and transportation mechanisms of pollutants entering lakes of varying characteristics affect lakes in different ways and to different degrees. This section gives a brief sampling of some of the negative effects discharges to lakes can have on the water, sediment, and biota. (For a more comprehensive explanation of the various effects of different pollutant discharges into lakes, see Reference 3.)

Water

Lake water can be degraded in numerous ways. For example, heavy nutrient loading from sewage, manure, and runoff from agricultural soils high in phosphorus reduce the amount of oxygen in the water. Moreover, various chemical compounds and metals entering the lake change the natural rates of chemical cycling in the lake by changing the ion concentrations in the lake. Acid rain forces calcium out of solution as manganese, zinc, and aluminum are brought in.

Sediments

Pollutant deposition into lake sediments can seriously alter existing water chemistry. Chemical pollutants deposited into the lake sediment alter the chemical balance and exchange of chemicals between the sediment and water. Sediment becomes a long-term storage site for pollutants. If sediment gets stirred up later on because of turnover, floodflows from tributaries, or dredging, even more pollutants are released into the water and the food chain and once again chemical exchanges between the lake bottom and the water are altered. For example, lead in sediments deposited up through the 1970s from vehicle exhaust is still found in lake sediments today. Altering water chemistry of a lake affects all biota in the lake.

Plants

Plants are vulnerable in several ways to pollutant discharge. If the discharge into the lake is sediment from agricultural, silvicultural, or construction activities, this new sediment layer can smother existing plants. Chemical pollutants can kill off some aquatic plants. Thermal pollution from some industrial plants or their cooling towers can kill plants by altering water temperature. Invasive species introduced by ballast water or by “hitchhiking” on pleasure boats hauled from one lake to another can wipe out plants by increased depredation. Acid rain reduces phytoplankton as well as zooplankton populations, undermining the aquatic food chain.

Fish

Fish are also very vulnerable to discharges into lakes. Some fish can lose their food source if aquatic plants are harmed or eliminated by pollutants. Thermal pollution from industrial cooling water affects fish metabolism and reproduction. It also stresses fish by increasing oxygen demand in the water and makes fish more vulnerable to disease. Reproduction can also be harmed by contact or ingestion of some chemicals. There might be fewer eggs, damaged eggs, or no eggs. Recently, estrogen in animal manure, human hormones, caffeine, and wood-pulp plant effluents are among some of the suspected causes of male fish producing eggs in their sex organs (4). Chemicals like chlorine are added in sewage treatment or flushed through cooling tower pipes to kill bacterial slimes and then are discharged into the lake and kill fish. Biomagnification in the food chain of substances like mercury, polychlorinated biphenyls (PCBs), or dioxins can pass from the sediments through phytoplankton and zooplankton into smaller and then bigger fish. High acid rain amounts not only wipe out entire fish species, but also cause lakes and streams to become fishless.

Fish can even serve as a transport mechanism for these chemicals from one region to another. For example, salmon have been found to contaminate lakes when they go upstream for spawning and then die in streams and lakes and release the PCBs or dioxins into the lake water and sediments. Beluga whales in the St. Lawrence River downstream of the Great Lakes have suffered impaired fertility and death from the DDT, PCBs, lead, mercury, and insecticides they have been exposed to from discharges

into the river as well as the Great Lakes. Once they have been dissected, their bodies must be disposed of according to hazardous waste regulations. Invasive species can harm fish. A northern snakehead, perhaps a former aquarium pet dumped by its owners, was recently caught in Chicago's Burnham Harbor on Lake Michigan (5). Snakeheads are considered "voracious eaters" and can consume many of the smaller fish in a lake. Invasive species may also start competing with existing fish by preying on the same food sources. Another exotic, the zebra mussel, affects fish by clearing the water and thus making fish more visible to prey.

Shellfish

Like fish, shellfish can also experience reduced reproduction from chemicals and hormones. For example, spawning behavior is altered by exposure to some antidepressants. Shellfish may face increased competition for food from invasive species. Shellfish also absorb and concentrate pollutants in their flesh, making them dangerous to eat.

Wildlife

Wildlife such as bear or cormorants can be harmed by biomagnification of PCBs, mercury, and so on when they eat the polluted fish.

Humans

Human populations are also harmed in numerous ways. Drinking water is degraded when a lake has industrial and agricultural chemicals, PPCPs, heavy metals, and so on in its source water. We know from painful experience how mercury causes Minamata disease or how cadmium, a byproduct of metal refinery, causes a disease that softens the bones. Algal blooms from cultural eutrophication give drinking water a bad taste and smell.

Shoreline use and value can also be diminished. Some lake pollution will force beaches to close, reducing recreational resources for locals and possibly keeping tourists away and out of the water because of *E. coli*, the smell and slime of algal blooms, and the toxins some blue-green algae produce. Heavy nutrient inputs into a lake can cause eutrophication that results in water that looks like pea soup and has a horrific smell. These can also cause a reduction in recreational use, when boats get caught up in algal mats or weedy macrophytes and tourists avoid an area, and lower the value of lakeshore properties. If fish or bear are unsafe to eat, then recreational opportunities are diminished. Tourists wanting to hunt and fish may go elsewhere, and those that continue to eat unsafe fish can harm their own health or have babies with birth defects.

PRESENT AND FUTURE CHOICES

Pollutant sources, transport, and fate have serious consequences for lakes. Cities will either have more degraded drinking water, with potential health consequences, or they will have to spend enormous amounts of money to purify the water enough to make it safe for human consumption. The effects of pollutants on aquatic life can not

only affect food chains within the lake but can also harm birds, wildlife, and humans that depend on the aquatic species for food. The externalities of industrial production, sewage or stormwater treatment, and other land uses are then spread to new populations as well as rebounding on polluter populations themselves. Their quality of life degrades along with their drinking water, recreational options, and fish and wildlife populations.

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LASERS SCAN LEVEES FROM THE AIR

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New Orleans District

U.S. Army Corps of Engineers

New Orleans District is taking to the air to learn more about the ground.

For good reason—for large jobs flying surveys are faster and cheaper than ground surveys. And the eye-in-the-sky is particularly handy in New Orleans District's 30,000 square miles of coastal Louisiana, where "ground" is a relative term.

The district is using laser technology called lidar, for LIght Detection And Ranging. It is analogous to radar, which stand for RAdio Detection And Ranging, except that lidar uses light beams instead of radio waves. A scanning laser rangefinder emits a beam of coherent light from a helicopter flying 45 miles per hour at an altitude of 220 feet. (The laser beam is invisible to the naked eye, and not intense enough to cause harm.) The beam bounces back and the lidar system's instruments gather vast amounts of data which can be sliced, diced, and served up in many forms. The process is almost effortless, compared to the labors of ground surveys in wetlands and bayous.

New Orleans District's first use of lidar was on the Lake Pontchartrain and Vicinity Hurricane Protection Project. Total cost was \$124,000, with most of the dollars going to the contractor, John E. Chance and Associated, Inc., which specializes in corridor mapping (electric lines, highways, and levees).

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A laser scanning rangefinder make a hefty instrument package for a helicopter (Photo courtesy of New Orleans District)

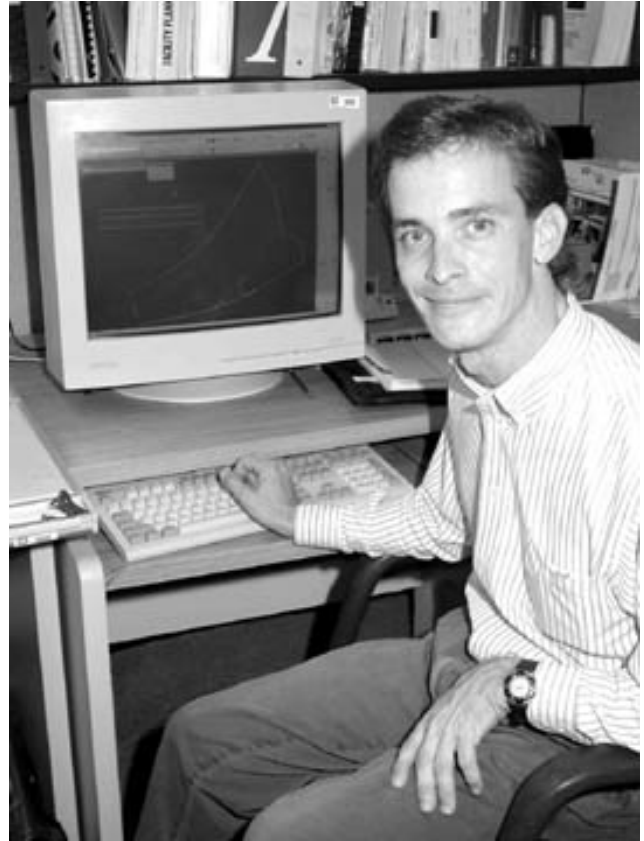
"We can rapidly determine levee heights without doing a ground survey," said Al Naomi, senior project manager of the hurricane project. "You can get a lot of data rapidly."

Indeed. The system produces 8,000 to 20,000 data points per second. But what does all this data really mean?

"It provides all three dimensions, the X, Y, and Z coordinates," said Terral Broussard, a district retiree who works for the contractor. This is trigonometry in action—three coordinates used to locate a point in space. The X and Y axis are length and width; the Z axis is the elevation.

Lidar surveys have been completed for all 125 miles of the Lake Pontchartrain and Vicinity project (www.mvn.usace.army.mil, click on hurricane box). These levees and floodwalls are in St. Bernard, Orleans, Jefferson, and St. Charles parishes. But lidar isn't limited to looking at levees. In New Orleans District, it is being considered for the Comite River Diversion Project, a flood control project to be built near Baton Rouge.

For land surveys, the laser is not the only sensor. Chance's helicopter also carries a video camera, and location is provided by the satellite-based Global Positioning System. The video provides high-resolution color images.



Mike Brennan, civil engineer, projects lidar information on his computer. (Photo courtesy of New Orleans District)

On the computer screen, these pictures may be viewed as an overlay precisely placed amid the lidar data. Or, you can just turn the video loose and watch the levee roll by as if you were on the helicopter.

To illustrate, Mark Huber of Survey Section pulled up on his computer a lidar aerial view of the Causeway toll plaza in Metairie, La. Bright colors marked the elevations. In mid-screen, he overlaid a video picture with the tollbooths, parked cars, roadways, and even the bike path that circles underneath, all clearly visible.

"Lidar may not be quite as accurate as ground surveys," Naomi said. "But it's faster, cheaper and requires no landowner interface."

"Other (Corps) districts have used lidar for land mapping," Huber said. "We're the first to pull the final product from the raw data, creating levee cross sections and centerline profiles."

How extensively New Orleans District will use lidar is unclear. While lidar cannot retrieve data from water surfaces, vegetation does reflect its signals. That makes it a candidate for wetland surveys where ground surveying is inherently difficult.

Lidar apparently has a lot of use as a quick look, and for studies. The question is whether it will prove accurate enough for project design. But, according to Huber, no errors exceed six inches compared with ground surveys.

LEVEES FOR FLOOD PROTECTION

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Levees were among the first river training structures built to protect human settlements and activities from floods. A levee is an embankment whose primary purpose is to furnish protection from high water due to river floods.

Levees are similar to small earth dams, but there are important differences:

- Levees are subject to high water levels only for a limited period of time (of the order of hours or rarely days), so that a levee embankment is generally not subject to saturation conditions.
- Levee alignment is dictated primarily by flood protection requirements; this often results in construction on poor foundations.
- Borrow is generally obtained from areas adjacent to the levee site, which produce fill material that is often far from ideal.

Embankments that are subject to water loading for prolonged time periods or permanently should be designed using earth dam criteria.

DESIGN OF LEVEES

Levee design involves numerous factors, dealing with both hydraulic and geotechnical considerations (Fig. 1). Design factors may vary, and no general procedure can be established. However, it is possible to present a general stepwise design procedure (1) based on successful experience that can be used as a starting point in levee design. Such a procedure may be summarized by the following steps:

1. Conduct a geologic study based on a thorough review of available data; initiate preliminary subsurface explorations.
2. Analyze preliminary exploration data and establish preliminary soil profiles, borrow locations, and embankment sections.
3. Initiate final exploration to provide

- additional information on soil profiles
- undisturbed strengths of foundation materials
- more detailed information on borrow areas and other required excavations.

4. Using the information obtained in step 3,
 - determine both embankment and foundation soil parameters, and refine preliminary sections where needed.
 - compute rough quantities of suitable material, and refine borrow area locations.
5. Divide the entire levee into reaches of similar foundation conditions, embankment height, and fill material, and assign a typical trial section to each reach.
6. Analyze each trial section for
 - foundation and embankment seepage;
 - slope stability;
 - settlement;
 - trafficability of the levee surface;
 - maintenance requirements.
7. Design special treatment to preclude any problems determined from step 6.
8. Based on the results of step 7, establish final sections for each reach, and define construction methods.
9. Design embankment slope protection.

Field Investigation and Laboratory Testing

Levee construction requires both field and laboratory investigation.

Field Investigation. The extent of field investigation depends on several factors (1). Conditions requiring extensive field investigation include poor experience in the area with respect to levee performance, significant levee height, bad foundation conditions, long high water duration, low quality of levee materials, high risk conditions in case of levee failure, and the presence of a concrete structure in the levee alignment.

Field investigation and testing are generally performed in three successive steps:

1. preliminary office geologic study;
2. field geologic survey;
3. field testing (preliminary phase and final phase).

Each step is described in Table 1.

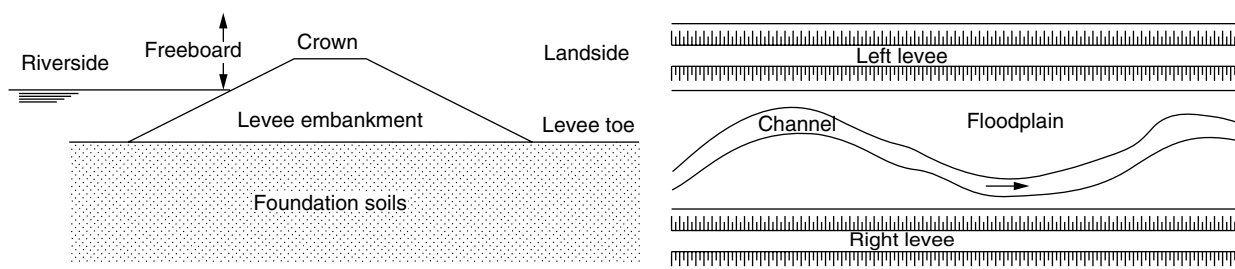


Figure 1. Sketch of a levee (cross section and plan view).

Table 1. Office Study and Field Investigation

Office study	Collection and study of — topographic, soil, and geologic maps — aerial photographs — existing information on soil and ground characteristics
Field survey	Observations and geologic analysis of the area, including such features as — riverbank slopes, rock outcrops, earth and rock cuts or fills — surface materials — poorly drained areas — evidence of instability of foundations and slopes — emerging seepage. — Physiographic features
Field testing	Preliminary phase — Widely spaced disturbed sample borings and test pits — Geophysical surveys or cone penetrometer test, used to interpolate between widely spaced borings — Borehole geophysical tests — Preliminary estimates of foundation strength Final phase — Additional disturbed sample borings — Undisturbed sample borings in critical reaches — Field vane shear tests, useful if undisturbed samples are unavailable. — Field pumping tests, providing information on large scale permeability of <i>in situ</i> deposits — Water table observations by piezometers in foundations and borrow areas

Laboratory Testing. Laboratory testing programs for levees vary from minimal to extensive, depending on the importance of the project and on the site conditions. Soil tests to be performed will vary according to the soil classification (Table 2).

Embankment Design

A levee cross section is generally trapezoidal berms may be added to ensure stability or seepage control. Berms are always used for high levee design.

Embankment design requires detailed analysis for levees of significant height, bad foundation conditions, or unsuitable embankment materials.

Embankment Geometry

Slope. Mechanical stability is largely dependent on the side slopes of the levee. The slope steepness is generally selected by considering the types of compaction, fill material characteristics, and foundation strength. For high levees, a soil mechanics stability analysis is required. For lower levees laid on proven foundations, recommended

riverbank slopes are about 1V:3H÷1V:3.5H; landside slopes are about 1V:2H÷1V:2.5H (V = Vertical; H = Horizontal).

Freeboard. Freeboard (the difference between the levee crown elevation and the high water stage) is used to account for uncertainties in levee performance. Many levees have been designed that have a 1-meter freeboard. Nowadays, the freeboard concept is less used in levee design. Risk-based analysis directly accounts for hydraulic uncertainties, and the geotechnical uncertainties are considered in the deterministic analysis of foundations and embankment behavior.

Crown Width. The width of the levee crown depends primarily on roadway requirement; it is in general in the range of 2.5–5 meters. To provide access for normal maintenance operations and flood fighting operations, minimum widths of 3.0–3.5 m are commonly used.

Stability Analysis. The principal mechanisms of levee failure are

Table 2. Laboratory Testing

Soil tests of fine-grained cohesive materials

- Visual classification and water content determination
- Atterberg limits
- Permeability
- Consolidation
- Compaction
- Shear strength

Soil tests of pervious materials

- Visual classification
- *In situ* density and relative density determination
- Granulometry
- Permeability
- Consolidation
- Shear strength

- overtopping
- piping
- embankment or foundation slides
- surface erosion.

Common methods used to analyze levee embankment stability are (2)

- methods assuming definite boundary stress values;
- methods based on the various soil characteristics that determine the failure state;
- methods considering the degree of safety for force and momentum equilibrium;
- methods comparing deformation work with the potential energy of the structure.

Primary attention should be devoted to the problems of defining the shear strengths, unit weights, geometry, and limits of possible sliding surfaces.

The more severe loading conditions for a levee are (1)

- End of construction. Levee embankment and foundation stability must be analyzed using drained conditions for free draining soils and undrained conditions for low permeability soils.
- Sudden drawdown. Excess pore water pressure can develop if a prolonged flood stage saturates the upstream embankment portion and then falls faster than the soil can drain. This can negatively affect riverside slope stability.
- Steady seepage from full flood stage. Flood water levels can sometimes fully saturate levee embankments. This is an uncommon condition for ordinary levees but may be critical for landside slope stability.

- Earthquake. Earthquake loadings are not normally considered in analyzing the stability of levees because of the low probability of coincidence with floods. Depending on the seismic characteristics of the study area, determination of liquefaction susceptibility may instead be required.

Shear strength selection under various loading conditions must consider soil type (free draining or low permeability) and pore water pressure conditions. Minimum factors of safety for the various stability analyses should be selected according to local regulations and to sound engineering judgement, considering failure consequences.

Measures to Increase Stability

Methods of Improving Embankment Stability. Levee stability can be enhanced by changes in embankment section, such as

Flattening Embankment Slopes. Flattening embankment slopes will usually increase the stability of an embankment against a shallow foundation type failure.

Stability Berms. Berms concentrate additional weight where it is most needed and force a substantial increase in the failure path. Berms can be an effective means of stabilization for shallow foundation and embankment type failures and also for more deeply seated foundation failures.

Methods of Improving Foundation Stability. Levees located on foundation soils of inadequate shear strength require some type of foundation treatment or the use of alternative design features in the levee section, such as a wider levee crest and flatter slopes. Foundation deposits that are prone to cause problems include very soft clays, sensitive clays, loose sands, natural organic deposits, and debris deposited by humans.

Means of dealing with inadequate foundation soils include

Excavation and Replacement. This procedure, though extremely effective, is economically feasible only where deposits of unsuitable material are not excessively deep.

Stage Construction. A levee is built in successive intervals of time where the strength of the foundation material is inadequate to support embankment weight, if built continuously at a pace faster than the foundation material can drain. Rest periods during levee construction permit dissipation of pore water pressures.

Stage construction is appropriate when pore water pressure dissipation is reasonably rapid. The rate of consolidation can be increased by using vertical drains. The disadvantages of this method are the delays in construction operations and uncertainty as to its scheduling and efficiency.

Densification of Loose Sands. Sands is generally not densified due to economic constrictions; possible exceptions are the presence of important structures in a levee system.

Levee Settlement. Evaluation of postconstruction settlement is important when settlement can result in loss of freeboard of the levee or damage to structures in the embankment. Detailed settlement analyses by theoretical analysis should be made when significant consolidation is expected, as under high embankment loads, embankments of highly compressible soil or built on compressible foundations, and near structures in levee systems founded on compressible soils.

Where foundation and embankment soils are pervious or semipervious, most of the settlement will occur during construction. For impervious soils, it is usually conservatively assumed that all the calculated settlement of a levee built in a normal sequence of construction operations will occur after construction.

Sometimes a levee is overbuilt by a given percentage of its height (varying from 0–10%) to take into account anticipated settlement of the foundation and within the levee fill itself, but overbuilding increases the severity of stability problems and may be impracticable for some foundations.

Seepage Control

Seepage flows develop when river waters flow through the embankment body or through foundation soils. Seepage is a very slow process, and the establishment of a seepage line generally requires a long time (2). As levees are exposed to high water levels for relatively short periods, the danger from seepage is more limited than that for earth dams.

Foundation Underseepage. Without control, underseepage flows in pervious foundations beneath levees may result in excessive pore water pressure, rupturing phenomena, and piping beneath the levee embankment. The principal seepage control measures are described in the following paragraphs (1). The effectiveness of seepage control measures can be evaluated by determination of flow-net, empirical methods, or mathematical solutions of seepage flow equations (Fig. 2).

Cutoffs. A cutoff beneath a levee stratum is one of the most positive means for blocking seepage through pervious soils. Cutoffs may consist of excavated trenches backfilled with compacted earth or slurry trenches usually located near the riverside toe. Steel sheet piling is generally not entirely watertight due to interlock leakage, but its use can significantly reduce the possibility of foundation piping. A cutoff must penetrate approximately 95% or more of the thickness of pervious strata to be effective, so constructing

it is not economically feasible in the presence of thick pervious strata.

Riverside Blankets. Levees are frequently situated in alluvial areas on foundations that have natural surface covers of fine-grained impervious to semipervious soils overlying pervious sand and gravel. If these surface strata (or blankets) are continuous and extend riverward for a considerable distance, they can effectively reduce seepage flow and pore pressures landside of the levee.

The effectiveness of the blanket depends on its thickness, length, distance to the levee riverside toe, and permeability. Protection of the riverside blanket against erosion is an important feature.

Landside Seepage Berms. Landside berms provide additional weight useful for counteracting uplift seepage forces. Berms also cause an increase in the hydraulic length of seepage flows, providing a reduction in pressure forces at the landside toe.

Seepage berms can improve the performance of existing impervious or semipervious top stratum but may also be placed directly on pervious deposits. Moreover, seepage berms increase the levee's landside slope stability.

Pervious Toe Trench. Partially penetrating toe trenches can improve seepage conditions close to the levee toe in a levee situated on deposits of pervious material overlain by little or no impervious material.

Pervious toe trenches are useful for controlling shallow underseepage. To collect deeper seepage flows, relief well systems may be used in conjunction with pervious toe trenches. Trench dimensions will vary according to the volume of expected underseepage, desired reduction in uplift pressure, and construction practicality. The trench backfill must be designed as a filter material. To improve drainage efficiency, trenches are often provided with perforated pipes to collect seepage.

Pressure Relief Wells. Pressure relief wells are used to reduce uplift seepage forces by providing controlled outlets for seepage that would otherwise emerge uncontrolled landward of the levee. They are installed where pervious strata underlying a levee are too deep to be penetrated by cutoff or toe drains and when the construction of landside seepage berms is not feasible due to space limitations.

The design of pressure relief well systems involves the determination of well spacing, size, and penetration in the underlying strata. Factors to be considered include

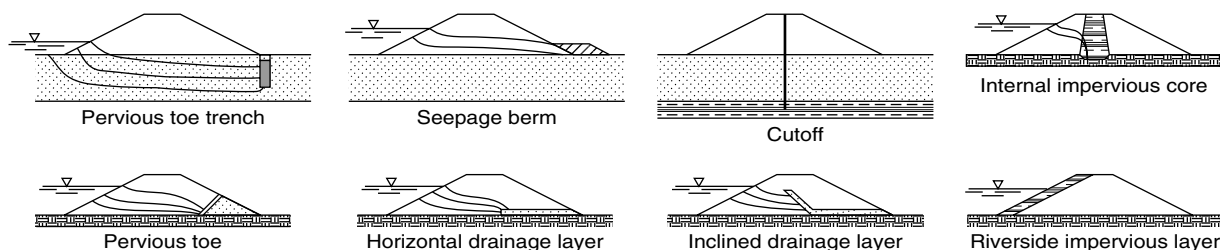


Figure 2. Seepage control devices.

depth, stratification and permeability of foundation soils, distance to the effective source of seepage, characteristics of the landside top stratum, and the degree of pressure relief desired.

Seepage Through Embankments. Embankment seepage emerging on the landside slope results in high seepage forces that decrease slope stability. Moreover seepage can soften fine-grained fill in the vicinity of the landside toe or lead to fine materials piping. Generally flood water stages do not act against a levee long enough to make seepage flows exit on the landside slope, but the combination of the effects of flood stages with a period of heavy precipitation may lead to consistent seepage flows. Various means to manage seepage through embankments are described hereafter.

Pervious Toe Drain. Pervious toe drains provide a ready exit for seepage through an embankment and can lower the phreatic surface sufficiently so that no seepage will emerge on the landside slope. They can also be combined with partially penetrating toe trenches as a method for controlling shallow underseepage.

Horizontal Drainage Layers. Horizontal drainage layers essentially serve the same purpose as pervious toes but can extend further under the embankment requiring a relatively small amount of additional material. They can also protect the base of the embankment against high uplift pressures where shallow foundation underseepage is occurring. Sometimes horizontal drainage layers also carry away seepage from shallow foundation drainage trenches.

Inclined Drainage Layers. An inclined drainage layer is very effective for controlling internal seepage and is used extensively in earth dams, but it is rarely used in levee construction because of its cost. It might find use for short levee reaches in urban areas. An inclined drainage layer completely intercepts embankment seepage regardless of the degree of stratification in the embankment or the material type riverward or landward of the drain. Inclined drains must be tied into horizontal drainage layers to provide an exit for the collected seepage.

The design of pervious toe drains and horizontal and inclined drainage layers must ensure that such drains have adequate thickness and permeability to transmit seepage without appreciable head loss and, at the same time, prevent migration of finer soil particles. Moreover, the design of drainage layers must satisfy the criteria for filter design.

Landside Berms. Landside berms can also be very useful in embankment seepage control. For a brief description of their characteristics, refer to the previous paragraphs.

Embankment Zoning. To counteract seepage flows, a central impervious core or covering a riverside slope with an impervious layer can be useful. The latter is generally more economical than a central impervious core and, in most cases, is entirely adequate. As a matter of fact, levee embankments are often constructed as homogeneous

sections as zoning is usually used in earth dams or in major levees.

PROTECTION OF RIVERSIDE SLOPES

The protection needed on a riverside slope to withstand the erosional forces of waves and stream currents depends on a number of factors such as floodwater stage duration, bank slope, erosion susceptibility of the embankment materials, presence of abrupt transitions, short-radius bends, or structures riverside of the levee that increase flow turbulence.

Several types of slope protection have been used. In general, high-class slope protection, such as riprap, articulated mat, soil cement, or paving should be provided on riverside slopes beneath bridges and adjacent to structures passing through levee embankments.

LEVEE CONSTRUCTION

Almost any soil is suitable for constructing levees, except very wet, fine-grained soils or highly organic soils. Levees of homogeneous cross section are generally made of silt, silty sand, or sandy clay. Fine-grained soils act as sealing materials due to their low permeability, but coarse components increase levee stability. The choice of material is often governed by the extent of local soils and by the technological experience of the contractor.

Levee embankments can be built with various degree of compaction. The central portion of the embankment may be compacted or semicompacted, but riverside and landside berms (for seepage or stability purposes) may be constructed of uncompacted or compacted fill. When foundations have adequate strength and where space is limited in urban areas both with respect to the quantity of borrow and levee geometry, compacted levee fill construction by earth dam procedures is frequently selected. This involves using select material, water content control, and compaction procedures. Where foundations are weak and compressible, high-quality fill construction is not justified because these foundations can support only levees with flat slopes. In such cases, uncompacted or semicompacted fill is appropriate. Uncompacted fill is generally used where the only available borrow is very wet and frequently has high organic content and where rainfall is very high during the construction season.

Except in seismically active areas or other areas requiring a high degree of compaction, compaction by vibration other than that afforded by tracked bulldozers is not generally necessary.

LEVEE REHABILITATION

As many levee systems are very old, the rehabilitation of existing levees is gaining greater prominence. Many techniques have been studied to rehabilitate existing levees. Recently, new working techniques that use geosynthetic materials extensively, have been introduced. Table 3 shows conventional and innovative levee rehabilitation techniques (3).

Table 3. Levee Rehabilitation Techniques

Problem	Conventional Rehabilitative Method	Innovative Rehabilitative Method
Overtopping protection	Rebuild Vegetation Concrete slabs	Cellular confinement system Reinforced grass Soil cement
Current and wave attack	Vegetation Revetment Gabions	Reinforced grass Concrete block system Soil cement
Rainfall surface erosion	Vegetation Chemical stabilization	Turf reinforcement mat
Embankment seepage	Toe drain Conventional chimney drain	Biopolymer chimney drain Biopolymer toe trench
Foundation seepage	Conventional toe trench Conventional cutoff Riverside blanket Landside seepage berm Pressure relief well	Jet grouted cutoff
Slope instability	Drainage Slope flattening and benching Conventional restraint structure Chemical treatment	Reinforced soil slope Soil nailing Pin pile Stone-filled trench Geosynthetic drainage system Anchored geosynthetic system

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LIMNOLOGY

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Limnology is the science of inland waters, also called epicontinental waters (1), including fresh and saline, standing (lenitic) and running (lotic) surface waters, estuaries, wetlands (2,3), and hyporheic (subterranean) waters (4) as ecosystems. Limnology integrates hydrological, physical, and chemical factors as framework constraints for ecological structures and processes. The first comprehensive publication coining the term "Limnology" was written by F. A. Forel (5) on Lake Geneva (Lac Léman) with chapters on geography, hydrography, geology, climatology, hydrology, hydraulics, thermics, optics, acoustics, chemistry, biology, history, navigation, and fisheries (see Ref. 6).

Limnology was traditionally focused on lake systems, as seen by the classic textbook of Hutchinson, *A Treatise on Limnology*, in Refs. 7–12. Other textbooks

on limnology that present lake systems in detail, running waters, however, are treated only in a few chapters (13–16). In 1922, the International Association of Theoretical and Applied Limnology (SIL) was founded, and limnology became the science of aquatic ecosystems. The volumes of the book series *Die Binnengewässer* (edited by A. Thienemann since 1925) present the basics of limnological knowledge. The first concepts to compare lakes (17–20) were directed to lake types, referring to size and morphology, climate, the behavior to stratify and mix seasonally or diurnally (21), and different nutrient contents resulting in different productivity and oxygen regimes (temperate, tropical and polar lakes, mono-, dimictic, and polymictic lakes, oligotrophic and eutrophic lakes, clear-water and dystrophic brown-water lakes). Regional limnology was included by Hutchinson (7) and (22) with comprehensive surveys over the lakes of the world, their mode of origin, size, and number.

Studies of hydrobiologists first focused on an inventory of plants and animals, and then on the microorganisms in standing and running waters. A more general concept of productivity and energy flow in food chains was developed to describe the input and fate of matter and energy in ecosystems (23). Lakes all over the world were investigated according to Lindeman's "Trophic-Dynamic Concept" on a large-scale comparison of the functioning of lake ecosystems (24), being a part of the International Biological Programme (IBP). The nonbiological fields of limnology were treated by physicists (25–27) and chemists (28,29) in basic reviews and the textbooks *Aquatic Chemistry* and *Physics and Chemistry of Lakes*.

Institutions for limnological research, all focusing on lake limnology, were founded during the early decades of the twentieth century in Austria, England, France,

Germany (including a first station on tropical limnology in Manaus, Brazil), Italy, the Netherlands, Sweden, Switzerland, Russia at Lake Baikal, the United States, and Canada. The only early stations for river ecology were at the Amazon and Volga rivers.

The limnology of creeks and rivers as part of aquatic ecology developed later than lake limnology (30,31). As running waters are more difficult to characterize, the development of concepts considered to suitably cover the relevant processes still proceeds. Typologies of rivers can be based on abiotic features as hydrological characteristics, water flow and its variability, the amount of dissolved and suspended matter, salt content and turbidity, and the course of temperature along the river and through time (32). *The Rivers Handbook* of Calow and Petts (33) has chapters on hydrology, physical and chemical characteristics, the biota, the ecosystem, and perturbations and biological impacts, and it addresses monitoring programs; modeling of hydrological development, of water quality, of sediment transport, and of biological responses; and the evolution of habitats.

First, in the early twentieth century fisheries, biologists divided European rivers along their stretch into distinct zones according to slopes, flow velocities, and the communities of fish (zones of salmonids, of cyprinids) (34). Benthos biologists amended this approach by including benthic invertebrates (rhitron, potamon) (35). In 1980, Vannote et al. (36) established the "River Continuum Concept" using the continuous shift along river stretches (1) of the relation between autotrophy and heterotrophy, and (2) of the relation between autochthonous nutrients, that are produced in the river, and allochthonous, terrestrial-borne nutrients. Rare or seasonal events as floods temporarily include the floodplains into the riverine system. The pulsed changes of dry and high-water conditions are essential for these systems adapted to regular floods, which affect the floodplains and offer new ecological niches (e.g., fruit-eating fish species in flooded Amazon forests) (37). The hereby derived "Flood Pulse Concept" sees the river as part of a larger ecosystem including the floodplains as system components being temporarily dry land or shallow flooded side-parts of the stream. The ecological continuity from water to land is approached by the "Concept of Ecotones" (38). The scale of integrated characteristics and management of the landscape results from enclosing the whole catchment area of the river system, based on the aquatic ecosystems, both rivers and lakes as partial systems. To characterize the structure and the hydrological processes within landscapes with respect to water flows, the integrated approach of the concept of "Ecohydrology" (39) was developed. The scale of catchment areas is the basis for the European Union (EU) "Water-Framework-Directive" that was established to keep or to reach a good/maximum chemical state and a good/maximum ecological state in all running waters and lakes of the EU-member states (40,41). Furthermore, special commissions have been established to deal with trans-boundary problems of water protection of rivers and lakes (Table 1).

Human use of land and waters changed the quality of inland waters with increasing population densities and by increasing loads of waste and nutrients impacting lakes and rivers. The limnology of creeks and rivers was predominantly developed within applied sciences, aiming at fisheries, water pollution by wastewater, the limits of self-purification of running waters, and solution of problems caused by pollution. Rivers have received urban wastewater for a long time. The pollution effects were first scientifically described at the beginning of the twentieth century. Kolkwitz and Marsson (42,43) characterized rivers from the source of pollution along stretches of self-purification by the benthic community's changes of species composition, being the base of the "Saprobic System." The saprobic system is still used as a biological tool to discriminate polysaprobic, mesosaprobic, and oligosaprobic conditions (44).

Wastewater pollution is a more severe threat for lakes than for rivers, because the organic loads, being nutrients for the aquatic organisms, remain in the lakes for years according to their water renewal time. The cumulative enrichment of nutrients, especially of phosphorus and nitrogen, with its biological consequences in lake waters (e.g., algal blooms) and the increasing of oxygen deficiencies in deep water layers, called "eutrophication," was observed in most lakes in densely settled areas of Europe and in North America. The increase of total phosphorus in lakes depends on the population density in the lake's catchment area. The limits for acceptable P-loadings of morphologically different lakes have been treated in comparative studies of the OECD in Europe and in Canada (45–48).

Systematic measures to reduce the inputs of P and N into surface waters have been taken for several decades (e.g., purification of wastewater in treatment plants, banning of detergents containing phosphate, abstraction of wastewater from lakes by ring-pipes). The results in many lakes clearly show that these efforts have been successful. The present state of nutrient loads of rivers and lakes in Europe and the development during the past is documented in reports, which can be downloaded from the European Environmental Agency (<http://www.eea.eu.int/>). Eutrophication of European lakes, in terms of phosphorus concentrations, generally is decreasing. The actual phosphorus concentrations are highest in the Eastern European countries and lowest in the Nordic countries. The state of surface waters in North America is documented in reports accessible from the U.S. Environmental Protection Agency (<http://www.epa.gov>).

Geological regions poor in carbonates are sensitive to acid rain as it changes the chemical composition of seeping runoff and ground- and surface-waters (49). Rain-acidified inland waters are carbonate-free, enriched in sulfate and aluminum, and acidic within a range between 4.5 and 5.5 pH units. Reports on long-term development and large-scale geographic distribution in EU-countries (50) showed that in Norway 6000 lakes (51) and in Sweden 17,000 lakes (52) were acidified. In southern Norway, 9630 fish populations were lost (53). Measures against acid rain and acidification of soils, lakes, and rivers started by reducing the emissions of acidic smoke (54) and direct treatment

Table 1. Commissions Established to Manage Inland Waters, River Basins, and Lake Catchments Including Trans-boundary Problems Within Their Catchment Areas. The Table Exemplarily Shows National and International Commissions and the Respective Lakes and Rivers

River, Lake, Basin	Commission	Web Page
Danube	International Commission for the Protection of the Danube River (ICPDR)	http://www.icpdr.org/pls/danubis/DANUBIS.navigators
Rhine	International Commission for the Protection of the Rhine River (IKSR)	http://www.iksr.org/
Elbe	International Commission for the Protection of the River Elbe (IKSE, SMOL)	http://www.ikse-mkol.de/html/ikse/ikse/deutsch/indexd.htm
Oder	International Commission for the Protection of the River Oder (signed by Germany, Czech Republic and Poland on 11 April 1996)	
Dnieper	International DNIPRO Fund (IDF)—National Program of Environmental Sanitation of River Dnipro Basin and Drinking Water Quality Improvement	http://greenfield.fortunecity.com/hunters/228/toppage1.htm
Bodensee, Lake Constance	International Commission for the Protection of the Lake Constance	http://www.igkb.de
Lake Geneva	Commission Internationale pour la Protection des Eaux du Lac Léman contre la pollution (CIPEL)	http://www.cipel.org
Great Lakes Basin	The International Joint Commission prevents and resolves disputes between the United States of America and Canada under the 1909 Boundary Waters Treaty. It assists the two countries in the protection of the trans-boundary environment, including the implementation of the Great Lakes Water Quality Agreement.	http://www.ijc.org http://www.epa.gov/glnpo
Lake Tahoe Lake Washington	Lahontan Regional Water Quality Control Board The Cedar River Basin Plan combines a traditional King County Basin Plan with a Non-point Source Pollution Action Plan. The Plan was prepared under the policy direction of the Watershed Management Committee composed of key tribal, state and local government agencies and non-governmental organizations, and a Citizens Advisory Committee of area residents.	http://www.epa.gov http://dnr.metrokc.gov/wlr/watersheds/cedar-lkwa.htm/
Mississippi River Basin	The Mississippi River Initiative began 1997 and has developed into a coordinated federal effort to keep illegal pollution, ranging from raw sewage to industrial waste, out of the river and to restore the river and surrounding communities. The Initiative employs cooperative efforts of the Dept. of Justice, EPA, Customs Service, Coast Guard, Fish and Wildlife Service, FBI, and institutions of 26 U.S. states of the Mississippi River Basin.	http://www.epa.gov
Aral Sea basin	Inter-State Commission for Water Coordination (ICWC): Water ministers of the five states in the basin, Kazakhstan, Kyrgyzstan, Turkmenistan, Tajikistan and Uzbekistan; the ICWC was established 1992 with joint responsibility for water management with the two river basin agencies (Amu-Darya and Syr-Darya).	

Note: See IWAC (www.iwac-riza.org) for complete list of trans-boundary co-operations in Europe.

of acidic waters. In Sweden, 200,000 tons of lime were spread every year over 20 years, and 6000 lakes were treated. About half of the acidified area was restored until the early 1990s (55–57).

Manmade lakes worldwide amount to tens of thousands of reservoirs and pit lakes from surface mining on coal, gravel, and ores. Reservoirs show limnological peculiarities (58,59) that primarily result from the management regime. The state of reservoir building and its problems are presented by the World Dam Commission (<http://www.wdc.org>). The water chemistry of pit lakes

from coal or ore mining is determined by geogenic impacts, mostly acidification and contamination by heavy metals. Reviews and other comprehensive scientific publications on mining lakes are in an initial state (60).

Some traditional institutions for limnology investigated “their” lakes over a long period. Eutrophication and acidification of surface waters by acid rain were recognized because of these long-term observations. The effects of countermeasures, stepwise reoligotrophication, and neutralization were also monitored hereby. Only long datasets by traditional long-term investigations (e.g.,

“Long Term Ecological Research”) render possible the analyses of the responses of lakes and rivers (61). Presently, these datasets are reevaluated to track the effects of long-term changes of climate on the physical, chemical, and ecological behavior of lake systems and rivers (62,63). In addition, paleolimnologists get information for previous changes of climate and of nutrient inputs altering in the course of historical and geological times, by investigating the fossil rests of organisms and other suitable proximate-indicators in laminated lake sediments (64–66). In most lakes of temperate regions, the sediments show data reaching back to the end of the last glaciation, and to some million years in the oldest, tertiary lakes like Lake Biwa, Japan (67).

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ADSORPTION OF METAL IONS ON BED SEDIMENTS

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The adsorption characteristics of cadmium on bed sediments of the River Ganga (India) have been studied to determine the tolerance of the river system for the heavy metal load. The effect of various controlling parameters, initial concentration, solution pH, sediment dose, contact time and particle size, have been evaluated.

The optimum contact time needed to reach equilibrium was approximately 60 minutes and was independent of the initial concentration of cadmium ions. The adsorption curves are smooth and continuous leading to saturation, suggesting a possible monolayer coverage of cadmium ions on the surface of the adsorbent. The extent of adsorption increases with pH. Furthermore, the adsorption of cadmium increases as adsorbent doses increase and as adsorbent particle size decreases. The two geochemical phases, iron and manganese oxides, probably support the adsorption of cadmium ions. The adsorption data have been analyzed by using the Langmuir and Freundlich adsorption models to determine the mechanistic parameters of the adsorption process. Isotherms have been used to obtain thermodynamic parameters, free energy change (ΔG°), enthalpy change (ΔH°) and entropy change (ΔS°). The negative values of free energy change (ΔG°) indicate the spontaneous nature of the adsorption of cadmium on bed sediments, and the positive values of enthalpy change (ΔH°) suggest the endothermic nature of the adsorption process. The uptake of cadmium is controlled by bulk as well as intraparticle diffusion mechanisms.

INTRODUCTION

Heavy metals added to a river system by natural and man-made sources during their transport are partitioned among different compartments of aquatic ecosystems, such as water, sediment, and biota (1). The added heavy metals then undergo many different chemical changes, whereby a high degree of variation in metal concentrations occurs (1,2). Metals that do not remain soluble in water are adsorbed and accumulate on bottom sediments that act as a sink (3). The distribution of heavy metals in the bottom sediments is affected by the mineralogical and chemical composition of the suspended material, anthropogenic influences, and *in situ* processes such as deposition, sorption, and enrichment in organisms (4). Thus, in the natural conditions of the river water, suspended and riverbed sediments play an important role in buffering higher metal concentrations in water, particularly by adsorption and/or precipitation. Therefore, the study of sediments and their sorptive properties can provide valuable information relating to the tolerance of the system to the added heavy metal load and may determine the fate and transport of pollutants in the aquatic environment.

The tremendous increase in the use of heavy metals during the past few decades has resulted in an increased concentration of metals in waterbodies and has great environmental significance due to their toxicity and nonbiodegradable nature (5). Cadmium, one of the most toxic metals, finds its way to watercourses through wastewater discharges from metal plating and cadmium–nickel battery industries, phosphate fertilizers, mining, pigments, stabilizers, and alloys (6). The toxic effects of cadmium include high blood pressure, kidney damage, and destruction of testicular tissue and red blood cells (7). Even low concentrations of cadmium

cause hypertensive diseases. The International Agency for Research on Cancer classified cadmium as one of the chemicals that is carcinogenic to humans (8). Therefore, understanding the environmental conditions and mechanisms that regulate the mobilization and distribution of cadmium in the environment is essential.

Gardiner (9) studied the effect of various parameters on the adsorption of cadmium on river mud and other natural solids. Koelmans and Lijklema (10) studied the adsorption of cadmium onto sediment and suspended solids in Lake Volkerak in The Netherlands and reported that cadmium is bound almost completely to geochemical iron, manganese, and organic phases. Wiley and Nelson (11) examined the influence of various factors on the adsorption of cadmium onto the sediments of Sturgeon Lake in Oregon and reported that pH is the most critical parameter affecting cadmium adsorption. Christensen (12) and Palheiros et al. (13) also reported almost similar findings. Bajracharya et al. (14) studied the effect of zinc and ammonium ions on the adsorption of cadmium on sand and soil and reported that both ions suppress adsorption capacity significantly. Fu and Allen (15) studied the adsorption of cadmium by oxic sediments using a multisite-binding model. The model has been used satisfactorily to predict the extent of adsorption across the pH range 4.5–7.0.

Despite the apparent wealth of information on adsorption processes, little is known about quantitatively describing adsorption by coarser sediment. The important components of the suspended load for geochemical transport are silt, clay, hydrous iron and manganese oxides, and organic matter. Generally, adsorption studies for sediment less than 50 μm in size predominate (13,16); those for sediment more than 75 μm in size are lacking because sorption decreases as particle size increase. Although clay and silt adsorb metal ions much better than coarser fractions of sediment, one should take into account that most river sediments contain 90–95% sand and only 0–10% clay and silt. Therefore, in river systems that have a high sand percentage and low clay and silt content, the overall contribution of the sand to adsorption of metal ions could be comparable to or even higher than that of the clay and silt fraction. Our earlier efforts to establish this hypothesis have been quite successful (17–19).

Recently, we studied the adsorption characteristics of zinc on bed sediments of the River Ganga (20). In this article, we report the adsorption of cadmium on bed sediments of the River Ganga to assess the tolerance of the river system to the added cadmium load. The importance of geochemical phases has been investigated. Adsorption data have been analyzed by using adsorption models to determine the mechanistic parameters of the adsorption process, and isotherms have been used to obtain thermodynamic parameters.

THE RIVER SYSTEM

The River Ganga is a perennial river formed by the confluence of two smaller rivers at Deoprayag. Bhagirathi is one of them originating at Gaumukh in the Gangotri Glacier, 3129 m above mean sea level; the other is

Alaknanda that originates in Sapta Tal Glacier. After covering a distance of about 220 km in the Himalayas, it enters the plains at Hardwar, and after meandering across a distance of about 2290 km in the plains of Uttar Pradesh, Bihar, and West Bengal, it joins the Bay of Bengal through a large number of branches flowing in India and Bangladesh (Fig.1).

Physiographically, the area is generally flat, except for the Siwalik Hills in the north and northeast of the catchment. The area is devoid of any relief features of provinces except from deep gorges cut by drains and rivers flowing through the area. The drainage pattern is dominated by the River Ganga, which is the only major river flowing through the area. The meltwater of the glaciers in the upper Himalayas maintains a perennial supply of water in the River Ganga.

The region is characterized by a moderate type of subtropical monsoonal climate. It has a cool, dry, winter season from October to March, a hot, dry, summer season from April to June, and a warm rainy season from July to September. The average annual rainfall across the Ganga Basin varies from 780 mm in the upper part to 1040 mm in the middle course and 1820 mm in the lower delta of Bangladesh (21). It has been observed that the rainfall is heaviest in the northern region of Hardwar district, close to the foothills of the Himalayas, and becomes less southward. The major land use is agriculture, and there is no effective forest cover.

Geologically, the area is a part of the west Indogangetic plain, which is composed mainly of Pleistocene and subrecent alluvium brought down by the river action from the Himalayan region. At Hardwar, the sequence of sandstone and shales along with gravel beds and clays are grouped within the upper and middle Siwaliks. The soils of this region do not form a compact block. They differ from valley to valley and slope to slope according to different ecological conditions. The bed of the river is rocky down to Hardwar. The soils of the area are predominantly loam to silty loam and are normally free from carbonates. The pollution status of the River Ganga has been described earlier (22,23).

EXPERIMENTAL METHODOLOGY

Freshly deposited sediments from shallow water near the bank of the River Ganga at Hardwar in the state of Uttaranchal were collected in polyethylene bags and brought to the laboratory. Samples were taken from the upper 5 cm of the sediments where flow rates were low and sedimentation was assumed (24,25).

The size distribution of the sediment samples was analyzed by using nylon sieves to obtain various fractions. The textural features of the sediments were observed, and a preliminary classification made according to grain size and distinctive geochemical features. The important geochemical phases for the adsorption process are organic matter, manganese oxides, iron oxides, and clays. The contents of manganese oxide and iron oxide were measured as total manganese and total iron, respectively, and were extracted from the sediment samples using acid

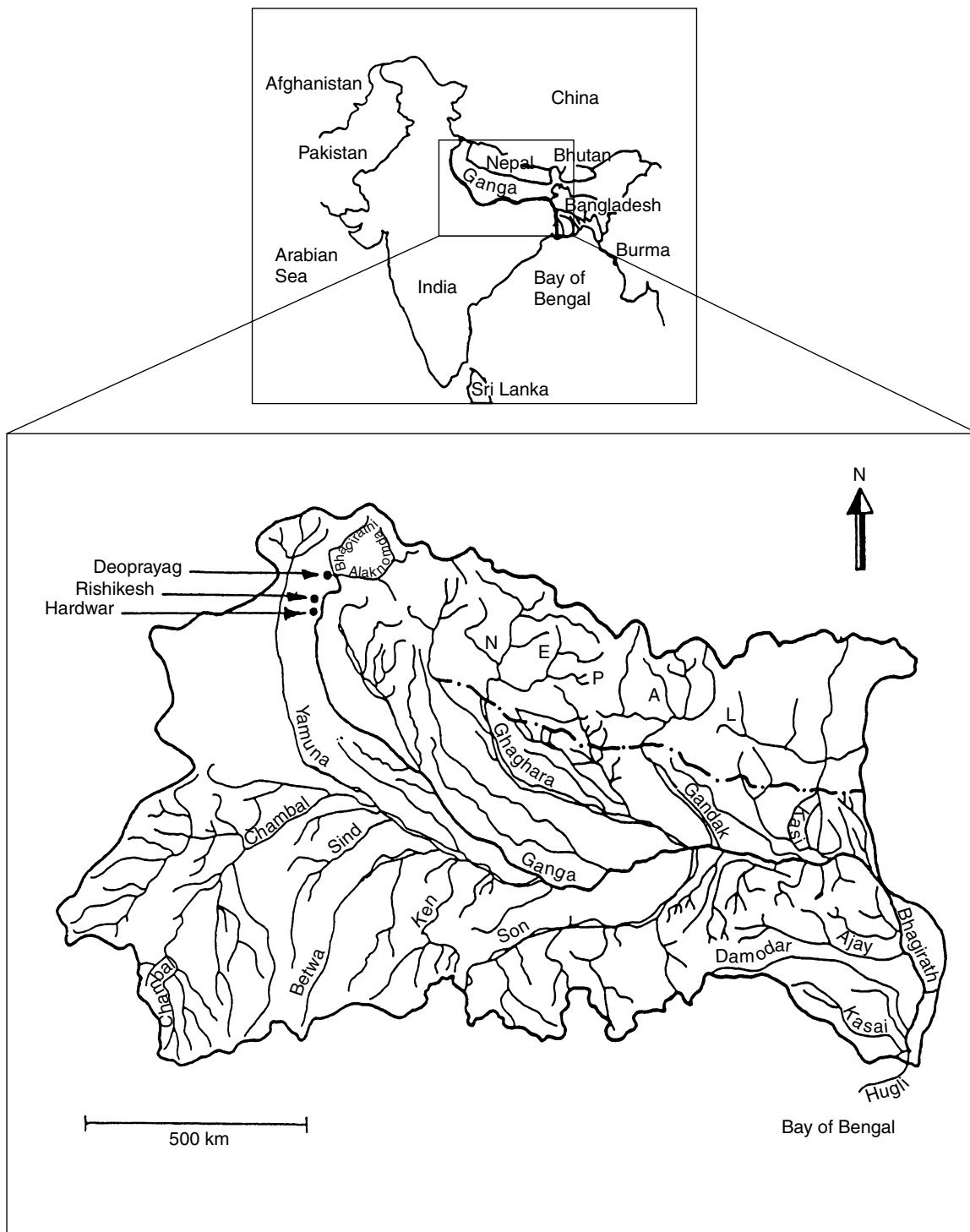


Figure 1. The Ganga Basin showing the locations of the sediment sampling sites.

digestion ($\text{HF} + \text{HClO}_3 + \text{HNO}_3$) in an open system. Organic matter was determined by oxidation with hydrogen peroxide.

All chemicals used in the study were obtained from Merck, India, and were of analytical grade. Aqueous solutions of cadmium were prepared from cadmium nitrate. Deionized water was used throughout the study. All glassware and other containers were thoroughly

cleaned by soaking in detergent followed by soaking in 10% nitric acid for 48 h and finally rinsed with deionized water several times prior to use.

Adsorption experiments were conducted in a series of Erlenmeyer flasks of 100 mL capacity covered with Teflon sheet to prevent introduction of any foreign particles. Fifty mL of cadmium ion solution ($200\text{--}2000\ \mu\text{g/L}$) was transferred in the flasks together with desired adsorbent

doses (W_s in g/L), and placed in a water bath shaker maintained at 30°C. A pH of 6.5 ± 0.1 was maintained throughout the experiment by using dilute HNO_3 and NaOH solutions. Aliquots were retrieved periodically and filtered through 0.45 μm cellulose nitrate membrane filters. The filters were soaked in 1% v/v HNO_3 for 1 h and thoroughly rinsed with deionized water prior to use.

The concentration of cadmium ions was determined by flame atomic absorption spectrometry, using a Perkin-Elmer Atomic Absorption Spectrometer (Model 3110) with air-acetylene flame. The detection limit for the cadmium ion was 0.0005 mg/L. Operational conditions were adjusted to yield optimal determinations. Quantification of metals was based on calibration curves of standard solutions of cadmium ion. These calibration curves were determined several times during the period of analysis.

RESULTS AND DISCUSSION

The content of important geochemical phases (iron and manganese) in different fractions of the sediment along with weight percentages are given in Table 1. The sediment has a coarse texture and is composed of more than 90% sediment of size $>75 \mu\text{m}$ and $<10\%$ silt and clay. The organic content of the sediment was of the order of 0–1%. The background cadmium level in the various fractions of the sediments was negligible in the unpolluted zone, compared to the amount of adsorbate added for the adsorption tests. This confirms the absence of any cadmium particulate attached to the sediment particles.

It is evident from the data that the amount of manganese and iron in the various fractions of the sediment decreases as particle size increases. This indicates the possibility that the two geochemical phases act as active support material for the adsorption of cadmium ions. However, the relative contribution of the individual components could not be obtained from the present studies because the type and composition of a sediment's mineral and organic fractions vary simultaneously in natural systems, and the effect of individual constituents cannot be isolated. The content of iron in the sediment fractions is relatively higher and indicates the possible presence of iron minerals other than hydroxides. However, this result should be confirmed by further investigations. It is further evident from Table 1 that the sediment fraction of 150–210 μm particle size constitutes 68.6% of the total sediment load. Therefore, it was considered appropriate to study the adsorption

of cadmium ions on this fraction (150–210 μm) and to compare it to the clay and silt fraction ($<75 \mu\text{m}$ particle size) to demonstrate the importance of the coarser fraction in controlling metal pollution.

OPERATING VARIABLES

Equilibrium Time (t)

To determine the equilibrium time for the adsorption process, adsorption experiments were performed for the uptake of cadmium ions for different contact times and for a fixed adsorbent dose of 0.5 g/L and an initial cadmium concentration of 1000 $\mu\text{g/L}$ for the two particle sizes of adsorbent (0–75 and 150–210 μm) at pH 6.5 (Fig. 2). The solution pH for the experiments was close to that encountered in the river water. These plots indicate that the concentration of cadmium ions in solution becomes asymptotic to the time axis such that there is no appreciable change in the remaining concentration after 60 minutes in both fractions. This time is presumed to represent the time at which an equilibrium concentration is attained. All additional experiments were conducted for 60 minutes.

According to Weber and Morris (26), the uptake for most adsorption processes varies almost proportionately as $t^{1/2}$ rather than as contact time. Therefore, plots of

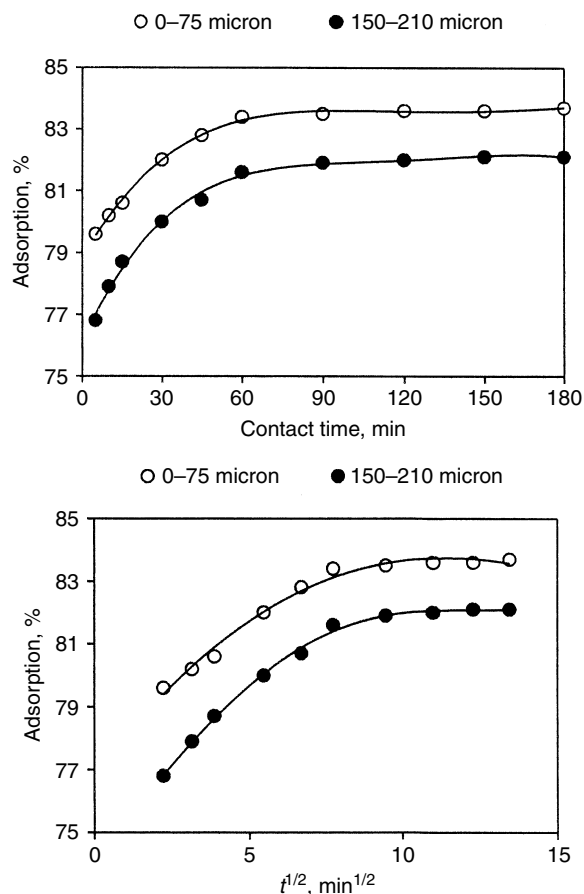


Figure 2. Effect of contact time (t and $t^{1/2}$) on percent adsorption of cadmium ions.

Table 1. Characteristics of Sediments

Sediment Fraction, μm	Weight, %	Total Mn, mg/g	Total Fe, mg/g
<75	9.8	1.570	68.2
75–150	13.5	1.420	62.0
150–210	68.6	1.199	58.0
210–250	6.6	0.844	55.1
250–300	0.7	0.830	52.1
300–425	0.6	0.823	52.0
425–600	0.2	0.824	52.0

cadmium adsorbed, C_t versus $t^{1/2}$, are also presented for the two particle sizes of adsorbent in Fig. 2. It is clearly evident that adsorption of cadmium ions follow three phases: (1) an early extremely fast uptake (Phase I), (2) a transition phase (Phase II), and (3) an almost flat plateau section (Phase III). Phase I is attributed to the instantaneous use of the most readily available adsorbing sites on the adsorbent surface (bulk diffusion). Phase II, exhibiting additional removal, is attributed to the diffusion of the adsorbate from the surface film into the macropores of the adsorbent (pore diffusion or intraparticle diffusion), stimulating further migration of adsorbate from the liquid phase onto the adsorbent surface. Phase III, a plateau section, represents the equilibrium state.

Adsorption Isotherm

The adsorption isotherms for cadmium adsorption on the bed sediments, shown in Fig. 3, have a fixed adsorbent dose of 0.5 g/L at a pH of 6.5 ± 0.1 . The adsorption data indicate linear distribution in the range 0 to 1000 $\mu\text{g/L}$. It is evident that for the same equilibration time, the amount of cadmium adsorbed is greater for greater initial concentrations of cadmium ions. In addition, the percentage adsorbed is greater for lower initial concentrations of cadmium and decreases as the initial concentration increases (Fig. 3). This is obvious because more efficient use of the adsorptive capacities of the adsorbent is expected from a greater driving force.

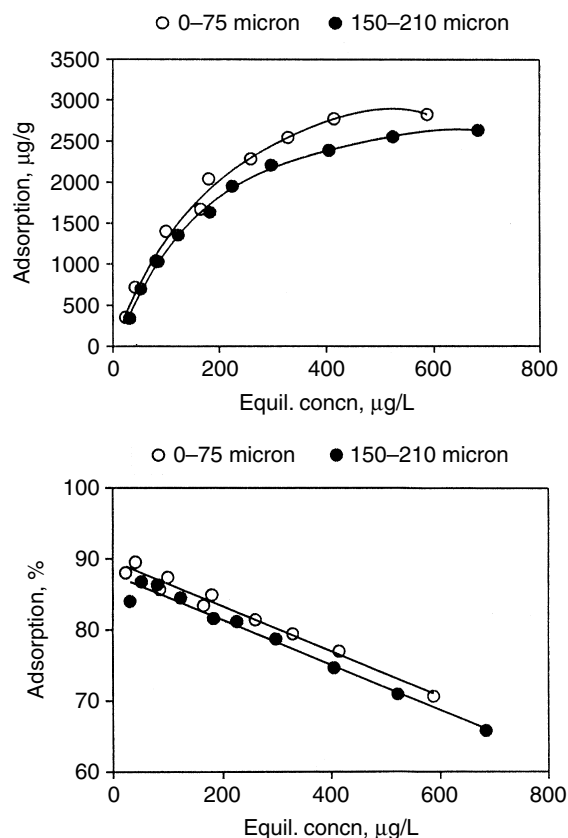


Figure 3. Adsorption of cadmium ions at different concentrations.

Comparing the two plots, it is clearly evident that the affinity of cadmium is greater for the $<75\text{-}\mu\text{m}$ fraction, clay and silt, compared to the coarser fraction. It is also evident from Table 1 that the $<75\text{ }\mu\text{m}$ fraction contains more iron and manganese than the 150–210 μm fraction, indicating the possibility of an association of these substrate (iron and manganese) with clay and silt particles. These findings illustrate the possible role of clay and silt as sites for cadmium adsorption. However, due to the paucity of data, correlation statistics could not be attempted to confirm this statement. The clay and silt constitute $<10\%$ of the total sediment load; comparing the weight percentages of the $<75\text{ }\mu\text{m}$ to the 150–210 μm and their corresponding adsorption capacities for cadmium ions, it is clear that the coarser sediment contributes more than the clay and silt fraction in controlling cadmium pollution.

Effect of pH

The adsorption of cadmium on the riverbed sediments was studied across the pH range 2–7 for a fixed initial concentration of cadmium ($C_i = 1000\text{ }\mu\text{g/L}$) and an adsorbent dose of 0.5 g/L at particle sizes of 0–75 and 150–210 μm (Fig. 4). The pH of the solution was adjusted by using dilute hydrochloric acid and sodium hydroxide solutions. The pH was measured before and after the solution had been in contact with the sediment; the difference between the two values was generally less than 0.1 pH unit. A general increase in adsorption as the pH of solution increased was observed up to pH 6.0 for both fractions of the sediment. From the results, it is evident that the pH for maximum uptake of cadmium ion is 6.0. Further, it is apparent that the adsorption of cadmium rises from 6.5% at pH 2.0 to 83.3% at pH 6.0 for the clay and silt fraction (0–75 μm) and from 3.9% at pH 2.0 to 81.5% at pH 6.0 for the coarser sediment fraction (150–210 μm). A similar trend for pH was reported by Palheiros et al. (13) for the adsorption of cadmium on riverbed sediment.

Adsorbent Dose (W_s)

The effect of adsorbent dose on the adsorption properties of the bed sediments of the River Ganga was studied at

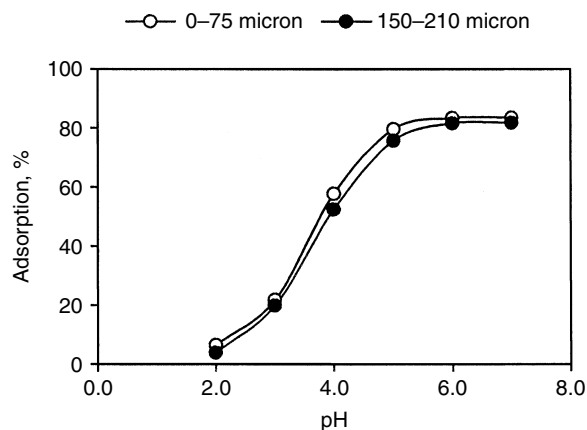


Figure 4. Effect of pH on adsorption of cadmium ions.

pH 6.5 using different adsorbent doses varying from 0.5 to 2.5 g/L and at a fixed initial cadmium concentration of 1000 $\mu\text{g/L}$ (Fig. 5). The experiments were conducted in a water bath shaker to disperse the sediment particles in the aqueous medium. It was observed that for a fixed initial concentration of cadmium ($C_0 = 1000 \mu\text{g/L}$), the adsorption of cadmium per unit weight of adsorbent decreases as adsorbent load increases. On the other hand, the percent adsorption increases from 83.4 to 93.7% for the 0–75 μm fraction as the adsorbent load increases from 0.5 to 2.5 g/L. The adsorption of cadmium was higher for the 0–75 μm fraction compared to the 150–210 μm fraction. Because of the higher surface area as well as higher content of iron and manganese in the 0–75 μm fraction; these are the main driving forces for the adsorption of cadmium ions.

Particle Size (d_p)

The effect of adsorbent particle size on cadmium adsorption is shown in Fig. 5 for a fixed initial concentration of cadmium ($C_i = 1000 \mu\text{g/L}$), an adsorbent dose of 0.5 g/L, and at pH 6.5. These plots reveal that for a fixed adsorbent dose, smaller particles adsorb more cadmium. Further, it is observed that the percentage of cadmium adsorbed decreases from 83.4% on the 0–75 μm fraction to 77.4% on the 425–600 μm fraction. This result occurs because

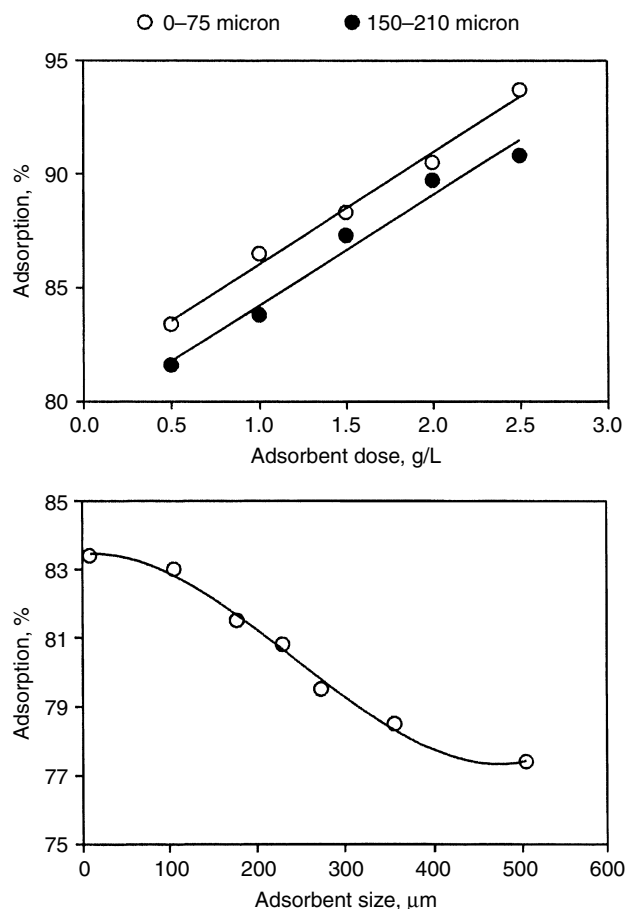


Figure 5. Effect of adsorbent dose and size on adsorption of cadmium ions.

adsorption is a surface phenomenon; the smaller particle sizes offered a comparatively larger surface area and hence higher adsorption occurred at equilibrium. The higher content of iron and manganese in the 0–75 μm sediment fraction also accounts for greater adsorption of cadmium in this fraction compared to larger size fractions of the sediment.

ADSORPTION MODELS

Adsorption data for a wide range of adsorbate concentrations are most conveniently described by adsorption models, such as the Langmuir or Freundlich isotherm, which relate adsorption density q_e (metal uptake per unit weight of adsorbent) to equilibrium adsorbate concentration in the bulk fluid phase, C_e . The adsorption data for cadmium on the bed sediments of the River Ganga was analyzed using Langmuir and Freundlich models to evaluate the mechanistic parameters of the adsorption process.

Langmuir Model

Langmuir's isotherm model is valid for monolayer adsorption onto a surface containing a finite number of identical sites. The Langmuir treatment is based on the assumption that maximum adsorption corresponds to a saturated monolayer of solute molecules on the adsorbent surface, that the energy of adsorption is constant, and that there is no transmigration of adsorbate in the plane of the surface.

The linear form of Langmuir isotherm equation is represented by the following equation:

$$\frac{1}{q_e} = \frac{1}{Q^0} + \frac{1}{bQ^0C_e} \quad (1)$$

where q_e is the amount adsorbed at equilibrium time ($\mu\text{g/g}$), C_e is the equilibrium concentration of the adsorbate ions ($\mu\text{g/L}$), and Q^0 and b are Langmuir constants related to maximum adsorptive capacity (monolayer capacity) and energy of adsorption, respectively. When $1/q_e$ is plotted against $1/C_e$, a straight line of slope $1/bQ^0$ and intercept $1/Q^0$ is obtained (Fig. 6), which shows that the adsorption of cadmium ions follows the Langmuir isotherm model. The Langmuir parameters, Q^0 and b , are calculated from the slope and intercept of the graphs and are given in Table 2. These values may be used for comparison and correlation of the sorptive properties of the sediments.

Freundlich Model

The Freundlich equation, widely used for many years, is applicable to isothermal adsorption. This is a special case of heterogeneous surface energies in which the energy term, b , in the Langmuir equation varies as a function of surface coverage, q_e , strictly due to variations in the heat of adsorption (27). The Freundlich equation has the general form,

$$q_e = K_F C_e^{1/n} \quad (2)$$

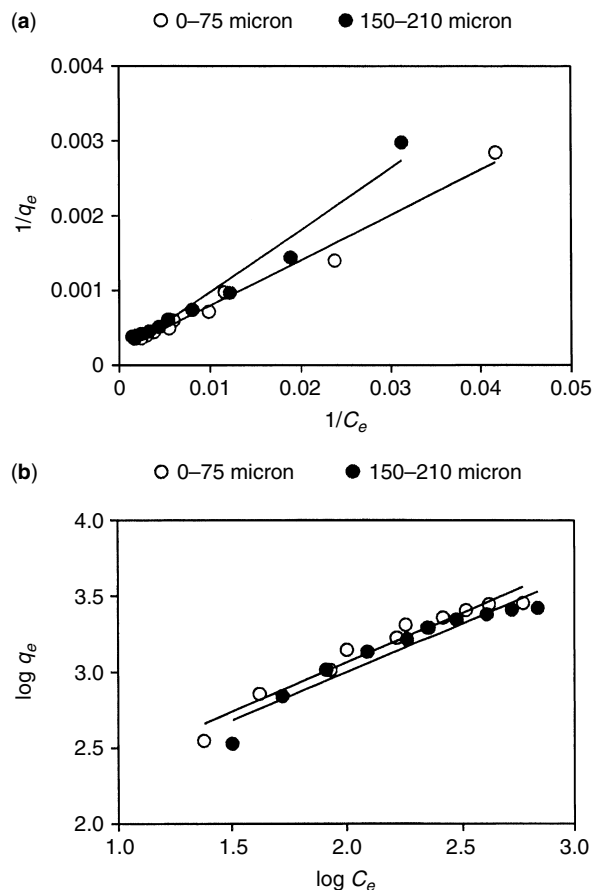


Figure 6. Graphical representation of adsorption isotherms: (a) Langmuir isotherm and (b) Freundlich isotherm.

The Freundlich equation is basically empirical but is often useful to describe data. Data are usually fitted to the logarithmic form of the equation,

$$\log q_e = \log K_F + \frac{1}{n} \log C_e \quad (3)$$

where q_e is the amount adsorbed ($\mu\text{g/g}$), C_e is the equilibrium concentration of the adsorbate ions ($\mu\text{g/L}$), and K_F and n are Freundlich constants related to adsorptive capacity and adsorptive intensity, respectively. When $\log q_e$ is plotted against $\log C_e$, a straight line of slope $1/n$ and intercept $\log K_F$ is obtained (Fig. 6). This plot reflects satisfaction of the Freundlich isotherm model for the adsorption of cadmium ions on Ganga River sediment. The intercept of the line, $\log K_F$, roughly indicates the adsorptive capacity, and the slope, $1/n$, indicates

Table 2. Langmuir Parameters for the Adsorption of Cadmium

Sediment Fraction, μm	Adsorption Maxima, Q° , mg/g	Binding Energy Constant, b , mg/L
0-75	5.0	3.306
150-210	10.0	1.206

Table 3. Freundlich Parameters for the Adsorption of Cadmium

Sediment Fraction, μm	Adsorptive Capacity, K_F mg/g	Adsorptive Intensity, $1/n$
0-75	0.0045	1.765
150-210	0.0043	1.726

adsorptive intensity (28). The Freundlich parameters for the adsorption of two metal ions are given in Table 3.

The Freundlich type adsorption isotherm is an indication of the surface heterogeneity of the adsorbent, and the Langmuir type isotherm corresponds to the surface homogeneity of the adsorbent. This leads to the conclusion that the surface of the bed sediments of the River Ganga is made up of small heterogeneous adsorptive patches that are similar to each other in respect to adsorptive phenomena (29).

THERMODYNAMICS

The effect of temperature on the adsorption of cadmium on bed sediments of the River Ganga is shown in Fig. 7 for a fixed adsorbent dose of 0.5 g/L and an initial cadmium concentration of 1000 $\mu\text{g/L}$ at pH 6.5 for the two sediment fractions (0-75 μm and 150-210 μm). The temperature range used in the study was from 20 to 40 $^\circ\text{C}$. It is evident from the plots that the adsorption of cadmium increases as temperature increases for both fractions, which may be mainly due to an increase in the number of active sites caused by the breaking of some bonds.

Thermodynamic parameters such as free energy change (ΔG°), enthalpy change (ΔH°), and entropy change (ΔS°) were determined by using the following equations (30-32).

$$\Delta G^\circ = -RT \ln K_c \quad (4)$$

$$\Delta G^\circ = \Delta H^\circ - T\Delta S^\circ \quad (5)$$

where

- ΔG° = change in free energy, kJ/mol;
- ΔH° = change in enthalpy, kJ/mol;
- ΔS° = change in entropy, J/mol/K;
- T = absolute temperature, K;
- R = gas constant = 8.314×10^{-3} ;
- K_c = equilibrium constant, defined as

$$K_c = \frac{C_{Ae}}{C_e} \quad (6)$$

where C_{Ae} and C_e are the equilibrium concentrations ($\mu\text{g/L}$) of the metal ions on the adsorbent and in the solution, respectively.

Equations 4 and 5 can be combined and rewritten as

$$\log K_c = \frac{\Delta S^\circ}{2.303R} - \frac{\Delta H^\circ}{2.303RT} \quad (7)$$

When $\log K_c$ is plotted against $1/T$, a straight line of slope $\frac{\Delta H^\circ}{2.303R}$ and intercept $\frac{\Delta S^\circ}{2.303R}$ is obtained (Fig. 8).

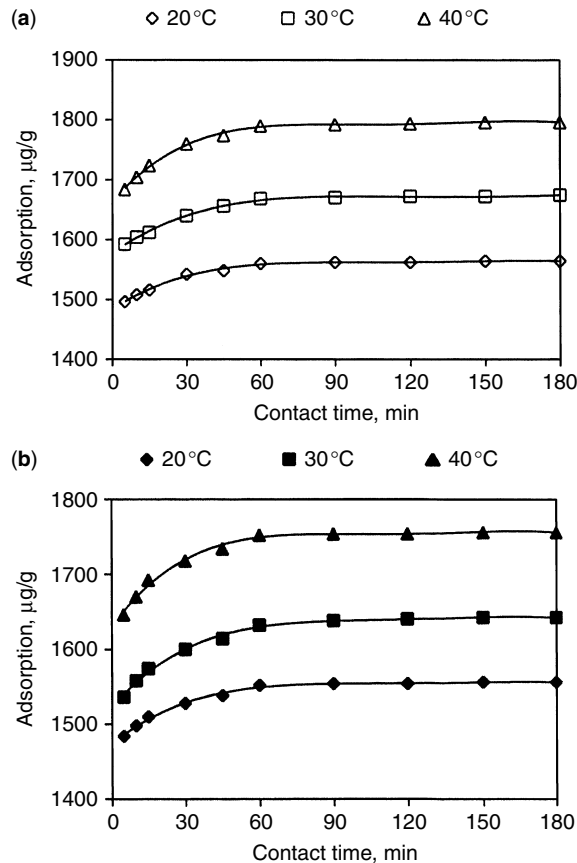


Figure 7. Effect of contact time and temperature on adsorption of cadmium ions: (a) sediment fraction = 0–75 μm ; (b) sediment fraction = 150–210 μm .

The values of ΔH° and ΔS° were obtained from the slope and intercept of the van't Hoff plots of $\log K_c$ vs $1/T$ (Fig. 8). The thermodynamic parameters for the adsorption process are given in Table 4.

Positive values of ΔH° suggest the endothermic nature of the adsorption. Negative values of ΔG° indicate the spontaneous nature of the adsorption process. However, the negative value of ΔG° decreased as temperature increased, indicating that the spontaneous nature of adsorption is inversely proportional to the temperature. The positive values of ΔS° show increased randomness at the solid/solution interface during the adsorption process. The adsorbed water molecules, which are displaced by the adsorbate species, gain more translational energy than is lost by the adsorbate ions, thus allowing the prevalence of randomness in the system. The enhancement of adsorptive capacity at higher temperatures may be attributed to the enlargement of pore size and/or activation of the adsorbent surface (33).

Adsorption Dynamics

The rate constant of adsorption is determined by using the Lagergren first-order rate expression, (34–36):

$$\log(q_e - q) = \log q_e - \frac{k_{ad}}{2.303} \times t \quad (8)$$

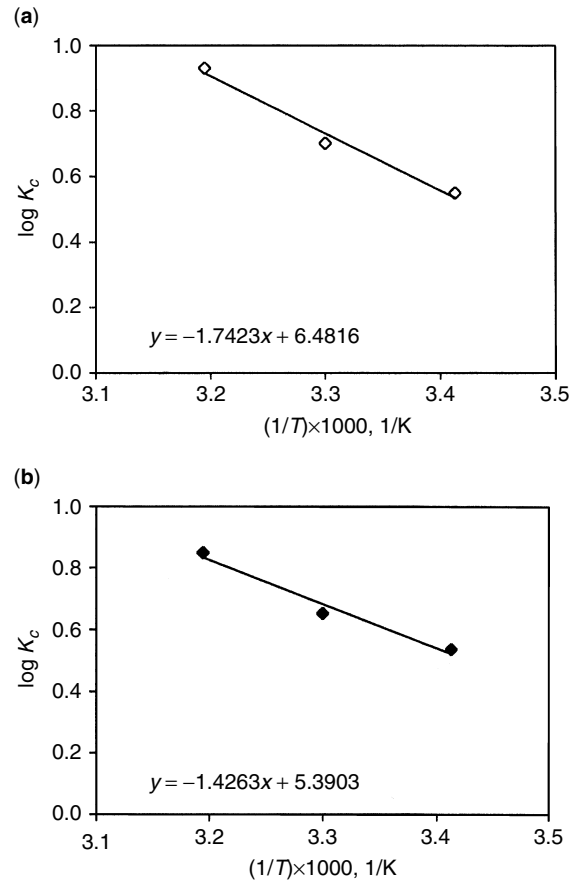


Figure 8. Van't Hoff plot for adsorption of cadmium ions: (a) sediment fraction = 0–75 μm ; (b) Sediment fraction = 150–210 μm .

Table 4. Thermodynamic Parameters for the Adsorption of Cadmium

Sediment Fraction, μm	Temperature, $^\circ\text{C}$	K_c	ΔG° , kJ/mol	ΔH° , kJ/mol	ΔS° , J/mol/K
0–75	20	3.545	–3.083	33.36	124.1
—	30	5.024	–4.066	—	—
—	40	8.524	–5.576	—	—
150–210	20	3.444	–3.012	27.31	103.2
—	30	4.494	–3.786	—	—
—	40	7.064	–5.087	—	—

where q and q_e are the amounts of metal adsorbed ($\mu\text{g/g}$) at time t (min) and at equilibrium, respectively, and k_{ad} is the Lagergren rate constant for adsorption ($1/\text{min}$). The straight line plots of $\log(q_e - q)$ versus t for different concentrations (Fig. 9) and temperatures (Fig. 10) indicate the applicability of this equation. Values of k_{ad} were calculated from the slope of the linear plots and are presented in Tables 5 and 6 for different concentrations and temperatures, respectively. The values of Lagergren rate constants (k_{ad}) were almost the same at both concentrations, whereas the value increases as temperature increases.

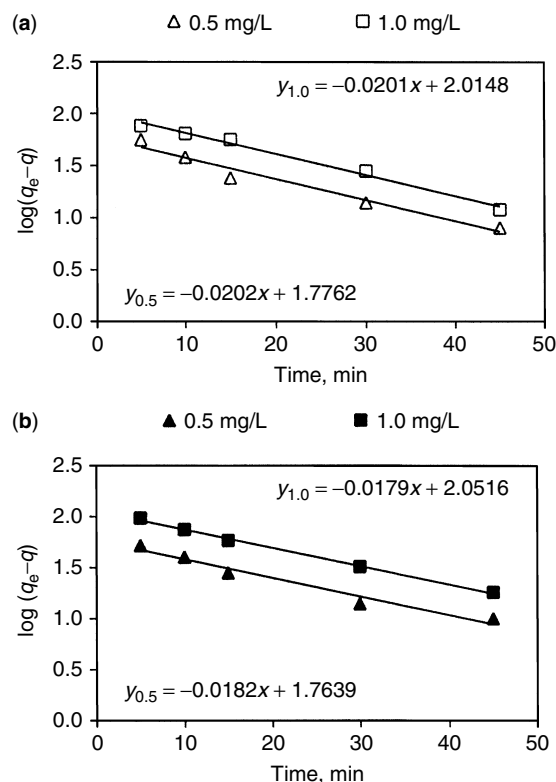


Figure 9. Lagergren plots at different initial concentrations: (a) sediment fraction = 0–75 μm ; (b) sediment fraction = 150–210 μm .

Intraparticle Diffusion

The rate constant for intraparticle diffusion (k_{id}) is given by Weber and Morris (37):

$$q = k_{id}t^{1/2} \quad (9)$$

where q is the amount adsorbed ($\mu\text{g/g}$) at time t (min). Plots of q versus $t^{1/2}$ are shown in Figs. 11 and 12 for different initial concentrations and temperatures, respectively. All plots have the same general features, an initial curved portion followed by a linear portion and a plateau. The initial curved portion is attributed to bulk diffusion, the linear portion to intraparticle diffusion, and the plateau to the equilibrium. This indicates that transport of cadmium ions from the solution through the particle solution interface, into the pores of the particle as well as the adsorption on the available surface of sediment, are both responsible for the uptake of cadmium ions. The deviation of the curves from the origin also indicates that intraparticle transport is not the only rate limiting step. The values of the rate constants (k_{id}) were obtained from the slope of the linear portion of the curves for each concentration of metal ions (Table 5) and temperature (Table 6). The values of intraparticle rate constants (k_{id}) increase as concentration and temperature increase.

CONCLUSION

The study has demonstrated that river sediments provide good buffering capacity for metal loads. Although cadmium

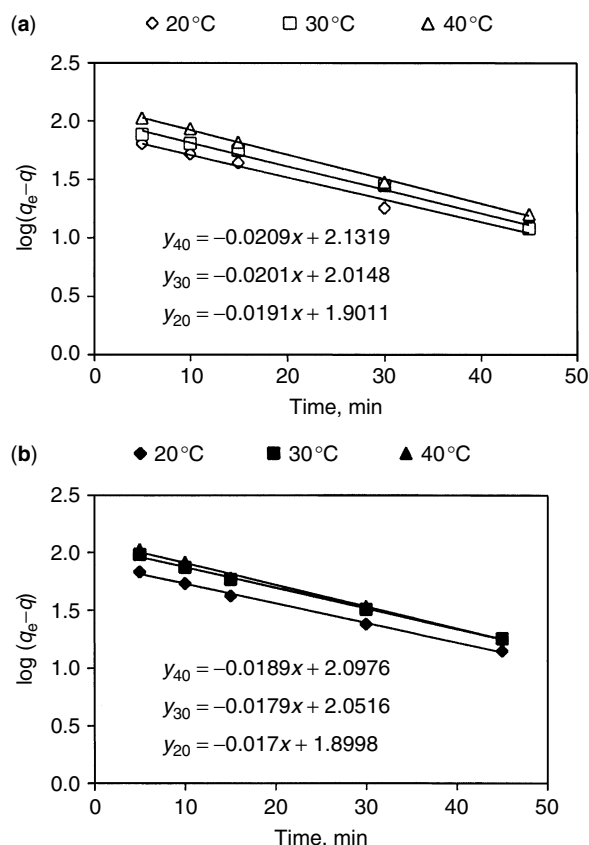


Figure 10. Lagergren plots at different temperatures: (a) sediment fraction = 0–75 μm ; (b) sediment fraction = 150–210 μm .

Table 5. Rate Constants at Different Concentrations

Sediment Fraction, μm	Concentration, $\mu\text{g/L}$	Lagergren Rate Constant k_{ad} (1/min)	Intraparticle Rate Constant k_{id} , $\mu\text{g/g/min}$
0–75	500	4.65×10^{-2}	3.84
—	1000	4.63×10^{-2}	9.88
150–210	500	4.19×10^{-2}	5.85
—	1000	4.12×10^{-2}	8.86

Table 6. Rate Constants at Different Temperatures

Sediment Fraction, μm	Temperature $^{\circ}\text{C}$	Lagergren Rate Constant k_{ad} (1/min)	Intraparticle Rate Constant k_{id} , $\mu\text{g/g/min}$
0–75	20	4.40×10^{-2}	8.11
—	30	4.63×10^{-2}	11.80
—	40	4.81×10^{-2}	14.29
150–210	20	3.92×10^{-2}	7.50
—	30	4.12×10^{-2}	10.00
—	40	4.35×10^{-2}	12.35

ions have more affinity for the clay and silt fraction of the sediment, the overall contribution of the coarser fraction to adsorption is greater compared to the clay and silt fraction. The pH of the solution is the most important parameter in the control of cadmium pollution.

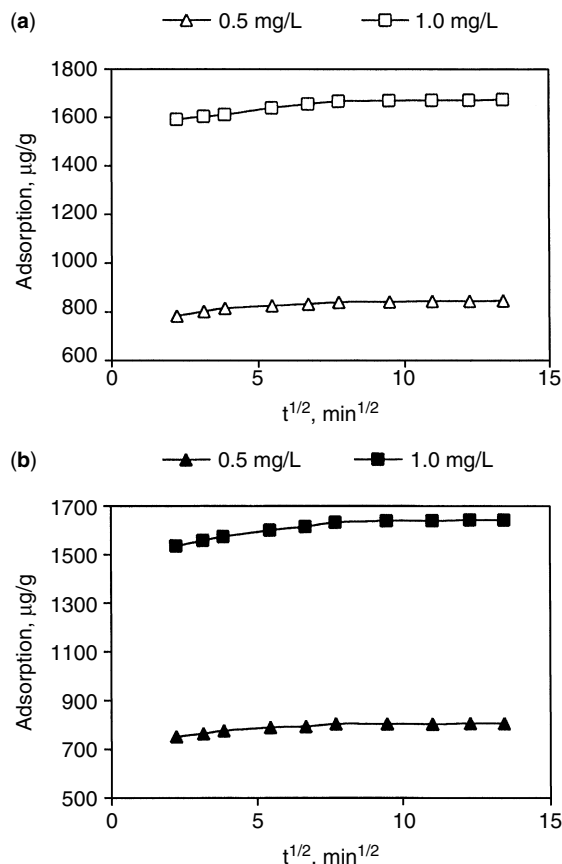


Figure 11. Intraparticle diffusion plots at different initial concentrations: (a) sediment fraction = 0–75 μm ; (b) sediment fraction = 150–210 μm .

Iron and manganese oxides support the adsorption of cadmium. The adsorption data follow both the Langmuir and Freundlich isotherm models. The kinetic data suggest that the adsorption of cadmium is an endothermic process, which is spontaneous at low temperature. The uptake of cadmium is controlled by both bulk as well as intraparticle diffusion mechanisms. It is stated that in natural river systems, salinity and pH greatly affect the speciation of the metal ions and thus greatly interfere with their fixation on clay minerals. Adsorption/desorption equilibria and complexation with fulvic and humic acids also play an important role in speciation. Further studies are being planned to understand better the processes that affect the sorption of cadmium in natural systems.

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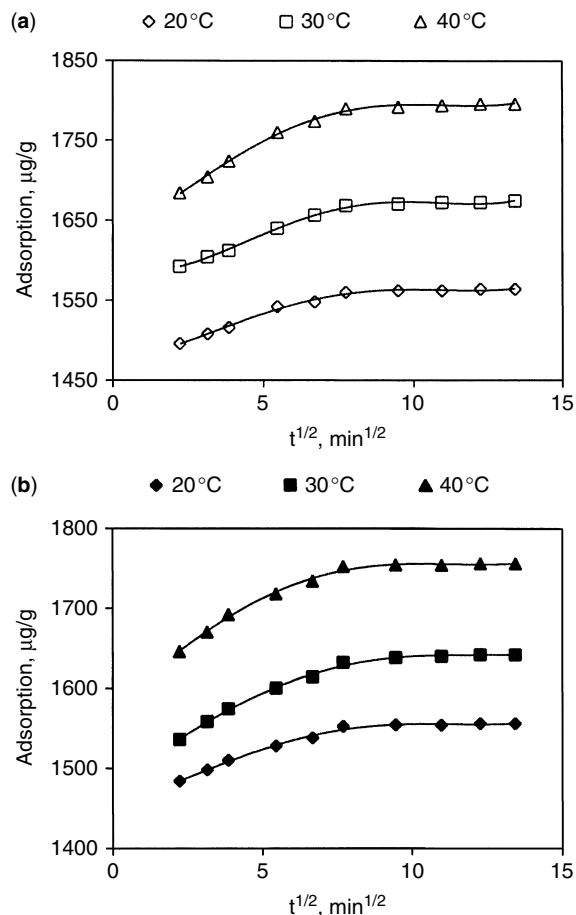


Figure 12. Intraparticle diffusion plots at different temperatures: (a) sediment fraction = 0–75 μm ; (b) sediment fraction = 150–210 μm .

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MICROBIOLOGY OF LOTIC AGGREGATES AND BIOFILMS

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INTRODUCTION

The microbial community is a vital component of lotic (flowing water) ecosystems and is important for nutrient cycling, organic matter decomposition, downstream transport of materials, and transfer of energy to higher trophic levels. Some of this microbial community is transported downstream as unattached cells; however, much of it is associated with suspended organic particles (i.e., aggregates) or attached to available substrate in biofilms. Research on microbial activity within organic aggregates has mostly been limited to marine and lake systems. As a result, more is known about the microbial communities and microbial processes in marine and lake aggregates compared with aggregates in lotic ecosystems. Conversely, biofilms have been studied in great detail in many systems, including lotic ecosystems. Many excellent and detailed reviews are available on the topics of aggregates and biofilms (1–11). Included here is a general overview of the microbiology and ecology of these topics and how they are relevant to lotic ecosystems.

LOTIC AGGREGATES

Origins and Terminology

Microorganisms readily colonize substrates in lotic ecosystems to form diverse and active communities. Suspended organic matter is often rich in nutrients and easily metabolized carbon, providing an especially good substrate for microbial colonization and growth and for the subsequent formation of lotic aggregates. Lotic aggregates can be defined as the interactive association between a microbial community and suspended organic matter in a flowing water system. The organic matter that is a central component of lotic aggregates may have a variety of origins including allochthonous and autochthonous particles and byproducts of animal and microbial activity. Allochthonous organic matter (e.g., wood, leaves, and

terrestrial debris) readily enters lotic systems through various pathways. Larger particles tend to quickly settle onto the channel bed or get trapped in channel structures, whereas the smallest particles often become entrained in the flowing water. Over time, physical and biological processes fragment the large particles into smaller detrital particles, which can then become resuspended into the water column. Organic matter particles of autochthonous origin (e.g., biofilm slough and aquatic vegetation) also may become fragmented and entrained in the current in the same manner as those particles of allochthonous origin. Organic byproducts of animal activity such as fecal pellets and discarded invertebrate exoskeletons are common in highly productive systems and can be nutrient-rich substrates for microbial colonization (12). Aggregates also may form in conjunction with dissolved organic matter (13) or colloidal organic polymers that are small (1.5 nm to 0.45 μm), high-molecular-weight byproducts of organic matter decomposition that remain suspended in the water (14).

Aggregates have the potential to enlarge as they travel downstream by combining with other aggregates (aggregation) and to gather additional materials primarily through impaction (Fig. 1) (15). However, the physical forces exerted by flowing and turbulent water usually fragment larger aggregates (disaggregation), maintaining the aggregates at a smaller size compared with those found in nonflowing systems (16). Although relatively small, the size of lotic aggregates are still highly variable and can be further classified into microaggregates (<5 to 500 μm) and macroaggregates (>500 μm) (7). Larger aggregates are also sometimes called “river snow” in keeping with terminology used to describe similar aggregates present in marine (marine snow) and lake (lake snow) ecosystems (17,18). In the field of stream ecology, lotic aggregates may also be called seston, which can be defined as particles entrained in flowing water. A list of common terms and abbreviations used to describe suspended matter in lotic systems is provided in Table 1.

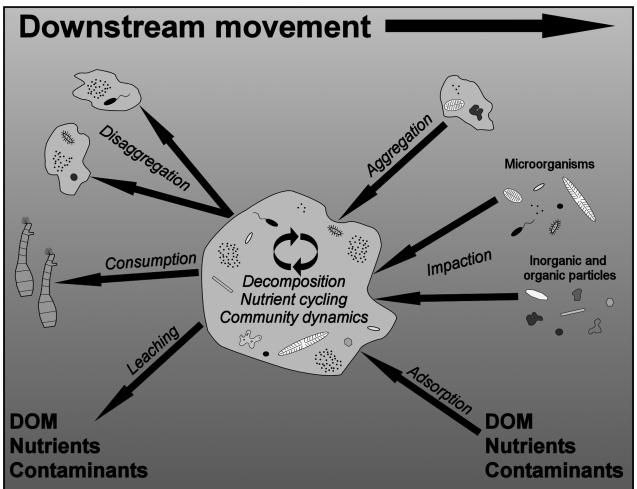


Figure 1. Conceptual diagram illustrating processes affecting lotic aggregate size and composition. DOM = dissolved organic matter.

Table 1. Common Terms, Abbreviations (in parentheses), and Sizes Used to Classify Suspended Matter in Flowing Water Systems

Term	Size
Dissolved organic matter (DOM)	<0.45 μm
Particulate organic matter (POM)	>0.45 μm
Course particulate organic matter (CPOM)	>1 mm
Fine particulate organic matter (FPOM)	0.45 μm to 1 mm
Aggregate	>0.45 μm
Macroaggregate	>500 μm
Microaggregate	<500 μm
Colloid	1.5 nm to 0.45 μm
Floc	>0.45 μm
Seston	>0.45 μm

The size ranges listed here are generally accepted definitions, but they can vary somewhat depending on application.

Microbial Community

Bacteria and other microorganisms secrete a viscous, slimy mixture of exopolymers (i.e., chains of polysaccharides) called glycocalyx. In general, the properties of exopolymers can contribute many beneficial functions to organisms including assisting in locomotion and adhesion to surfaces, and protection from abrasion, dehydration, and nutrient loss. More specifically, in lotic aggregates, the exopolymers bind the contents of aggregates together and assist in the capture of additional nutrients, organic matter, and other microorganisms. Much of the exopolymers associated with lotic aggregates are likely produced by microorganisms within the aggregate, but they also may accumulate via impaction with suspended exopolymers.

Microbial communities thriving within the fragile matrix of exopolymers and particles consist of a diverse heterogeneous assemblage of bacteria, algae, fungi, protozoa, and sometimes zooplankton. Compared with marine and lake ecosystems, relatively few studies have examined the microbial community structure of aggregates in lotic systems. However, many studies on marine aggregates have demonstrated that the concentration of algae, zooplankton, protozoa, and bacteria within aggregates is much greater, often 2–3 orders of magnitude, than in the surrounding water (7). Larger aggregates often support larger microbial communities, but smaller particles may be relatively more densely populated per unit surface area (19). Detailed microscopic analyses have shown that bacteria are not uniformly distributed within or on the surface of aggregates but form microcolonies (7). These microbial communities have temporally dynamic compositions that change quickly in response to conditions both inside and outside of the aggregate and can also exhibit seasonal patterns (17,20).

Inorganic (nitrogen, phosphorus, and silica) and organic (carbohydrates, amino acids, and dissolved organic carbon) nutrients are often 1–2 orders of magnitude more enriched in aggregates compared with surrounding water (7). In addition, many heavy metal and organic contaminants also readily adsorb to the aggregate matrix. As a result of these enriched conditions, downstream movement of aggregates is an important mechanism of transport for nutrients, organic materials, and contaminants. Moreover,

these nutrient-rich substrates result in high growth efficiencies of heterotrophic bacteria associated with the lotic aggregates, which suggests that these aggregates are hot spots for microbial processes and the transfer of organic matter into bacterial biomass (20). Enzymes produced by aggregate-bound bacteria solubilize organic matter 1–2 orders of magnitude faster than the bacterial cells can assimilate the dissolved matter (21). This decomposition results in aggregate mass loss and leaching of dissolved organic matter back into the flowing water. Finally, consumption of lotic aggregates by many species of detritus- and filter-feeding organisms is a vital link of energy transfer between microbial and higher trophic levels (Fig. 1).

BIOFILMS

Biofilm Structure

Microbial biofilms are the thin slime layers that are observed on surfaces and substrates (e.g., rocks, sediment, vegetation, leaves, wood) in aquatic systems. Biofilms (sometimes called microbial mats) are similar to aggregates and possess many of the same general structural and functional attributes. For example, biofilms (like aggregates) contain microbial communities that are vital to lotic ecosystem function and are responsible for many key production, decomposition, and nutrient cycling processes (22). Highly developed biofilms are generally a few millimeters thick and can develop on virtually any available surfaces in natural lotic systems, provided the system has sufficient nutrients. Biofilm development is especially rapid on surfaces that contain or leach nutrients (e.g., particulate organic matter, POM) (2). Available surfaces are still colonized in nutrient-poor or oligotrophic systems, but biofilms are generally less developed. Under oligotrophic conditions, biofilm bacteria are often able to obtain limited nutrients by concentrating organics on the exopolymer matrix, using waste products of nearby microorganisms, and by pooling their biochemical resources (i.e., enzymes) to break down organic particles (11).

Historically, biofilms were thought to be a relatively homogeneous distribution of bacteria in an exopolymer matrix and nutrient and material delivery throughout the biofilm would be dominated by diffusion (2). However, recent advances in the analytical tools used to study biofilms have allowed scientists to more accurately characterize the structure or architecture of biofilms. For example, the application of confocal scanning laser microscopy has allowed nondestructive inspection of intact biofilm samples. This technique has revealed the physical structure of established biofilms to consist of a diverse microbial assemblage arranged in microcolonies and interspersed within a heterogeneous exopolymer matrix containing protrusions and a network of pores and channels (2,4). Microsensors, another relatively recent research tool, have been used to examine chemical properties in spatially explicit areas within biofilms and have found distinct differences in dissolved oxygen concentrations between the pores and channels in the

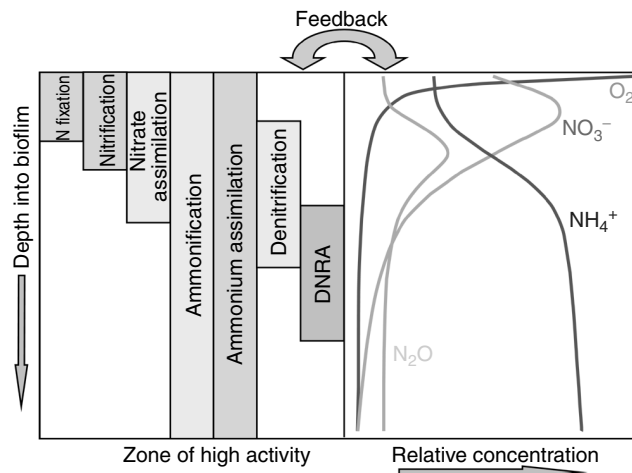


Figure 2. General patterns of nitrogen cycling (left) and relative concentrations of nitrogen constituents (right) in a lotic biofilm [redrawn and modified from Paerl and Pinckney (3)]. Feedback between the chemical and biological components facilitates the formation and equilibrium of the gradient (i.e., the chemical environment within the biofilm affects microbial processes and vice versa). DNRA = dissimilatory nitrate reduction to ammonium.

matrix and the matrix itself. These differences suggest that such structures may facilitate water movement and nutrient delivery within the biofilm. Microsensors have also been used to quantify vertical chemical gradients in biofilms that have important implications for microbial community structure and biogeochemical processing (Fig. 2). Chemical gradients will be discussed further in the next section.

The physical stress caused by the turbulent flow of water in lotic systems can have significant effects on biofilm structure and organization. Many studies have observed that biofilms subjected to fast flowing water tend to be thinner, smoother, denser, more stable, and have lower surface sinuosity than biofilms in slower or nonflowing water (23,24). Cells occupying the surface of lotic biofilms have been observed to be arranged in formations that can withstand the higher shear stress; examples of these formations include hexagon-shaped structures and ripples positioned parallel to flow direction (24). Freely oscillating streamer-like structures are also sometimes observed adhered to the surface of lotic biofilms (8). Streamers are predominately single species colonies that elongate over time in the absence of disturbance.

Microbial Processes

The rates and types of microbial processes at work in any particular biofilm are largely regulated by the physical and chemical characteristics of the lotic environment. Several types of algae and cyanobacteria flourish in biofilms if sufficient nutrients are available and the overlying water is shallow and clear enough to allow adequate light penetration to the substrate. Oxygen (O₂) generated by these primary producers can reach very high levels and is a key factor supporting aerobic biofilm processes. In

fact, under high-light conditions, the surficial area of well-lit biofilms can produce O_2 concentrations 2–3 times greater than atmospheric concentrations (25). Much, if not all, of the oxygen is respired as it is transported and diffused deeper into the biofilm creating a vertical gradient of oxygen concentrations throughout the biofilm (Fig. 2). Chemical gradients are common in biofilms and control which microbial processes are able to occur at certain depths. The opposite is also true; microbial processes control the availability of certain ions. The feedback relationship between these two factors quickly forms a relatively stable gradient equilibrium that is particularly important for the transformations and cycling of certain nutrients, primarily nitrogen, carbon, and sulfur.

The nitrogen cycle is a complex group of microbial processes that transform, oxidize, or reduce nitrogen ions among the various oxidation states. These processes occur under different environmental conditions, all of which can be located in distinct locations along the chemical gradients found in biofilms (Fig. 2). Nitrogen fixation, which is the conversion of atmospheric dinitrogen gas (N_2) to biologically available ammonium (NH_4^+), is predominately done by cyanobacteria; thus, it occurs in the upper, high-light areas of the biofilm. Nitrification, which is the oxidation of NH_4^+ to nitrate (NO_3^-), also occurs in the upper areas because it is an aerobic process. Bacteria that have the enzymes to reduce NO_3^- for assimilation likely occur throughout the biofilm profile, but they can only use NO_3^- where it is present. Organic matter is ubiquitous throughout the biofilm; therefore, ammonification (the release of NH_4^+ from organic matter) and the resulting NH_4^+ are also widespread. Ammonium assimilation into microbial biomass is not oxygen-dependent and can take place wherever NH_4^+ is present. Denitrification, which is the anaerobic reduction of NO_3^- to nitrite (NO_2^-), nitrous oxide (N_2O), and N_2 , occurs deeper in the biofilm where oxygen is low and NO_3^- is high. Dissimilatory NO_3^- reduction to NH_4^+ is strictly an

anaerobic process that occurs in the lowest areas of the biofilm where NO_3^- is still available.

It is generally accepted that bacterial activity and production of biofilm biomass is resource limited in lotic systems. Thus, nutrient availability and the quantity and quality of the dissolved organic carbon in the water have strong positive effects on bacterial activity in the biofilm (26,27). Total bacterial activity also often correlates well with primary production in biofilms because of the influx of oxygen and high-quality organic carbon provided by algae (27,28). Finally, vertical hydrodynamics of stream channels (i.e., upwelling and downwelling) can be an important means of nutrient and dissolved organic matter (DOM) delivery through the biofilm, which can subsequently increase microbial activity (29).

Under certain chemical or physical conditions, specialized biofilms that are dominated by unique groups of microorganisms may develop. For example, orange-colored iron-oxidizing biofilms are prevalent in areas where groundwater with high concentrations of reduced iron (Fe^{2+}) is discharged into oxygenated waters (30). Another example is the colorful biofilms of thermophilic bacteria and cyanobacteria that are commonly associated with water flowing from hot springs (Fig. 3). The colors of these biofilms depend primarily on the chlorophyll content in the cells, but also on the ratio of chlorophyll to carotenoids (yellow to red pigments) (31).

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Figure 3. Orange-colored biofilm associated with hot spring outflow draining into the Firehole River, Yellowstone National Park, Wyoming. The bright orange color is because of pigments in thermophilic cyanobacteria and bacteria in the biofilm.

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MICROORGANISMS IN THEIR NATURAL ENVIRONMENT

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GENERAL REMARKS

In their natural aquatic environment, microbes face conditions that principally differ from the picture of a mixture of different microbes swimming around in a kind of liquid culture. The study of microorganisms in laboratory cultures often gives limited information about the role and behavior of microorganisms in their natural habitat. Our understanding of microbes in their natural aquatic habitat was considerably increased by the development of *in situ* methods in the last decades. The introduction of microsensors allowed characterizing the microbial habitat on the size scale of microorganisms. The recent advances in microscopic techniques (confocal laser scanning microscopy, atomic force microscopy) offer the possibility of real 3-D examination of microbial aggregates or biofilms. Molecular techniques such as *in situ* hybridization to study the phylogeny and physiology of microorganisms in their natural environment are rapidly improving.

A characteristic of the natural habitat is its heterogeneity. Hot spots of high activity may be surrounded by a "desert." Natural habitats are characterized by **gradients**. The co-occurrence of various, sometimes opposing, gradients creates a high diversity of ecological niches.

Depending on transport velocities and reaction rates, these gradients can occur on a meter scale (in open water) down to a micrometer (in sediments) or even submicrometer scale (in microbial aggregates) (Fig. 1). The natural environment is not static but subject to fast **temporal fluctuations**. Temporal fluctuations occur on different timescales from seasonal effects down to very short-term changes within seconds or minutes. A phototrophic microbial mat, for example, can shift from oxygen oversaturation to anoxia within minutes when it is darkened (1). Microorganisms have developed different strategies to cope with changing conditions. The largest part of the bacteria in aquatic systems is in a metabolically inactive dormant state, waiting for favorable conditions. Motility, often in combination with chemotaxis, is an important strategy for following changing gradients. Microorganisms can also alter their phenotypic properties in response to changing conditions. On the scale of microorganisms, surface effects are important and **diffusion** becomes the dominant transport process. Solid surfaces are covered by a diffusive boundary layer (DBL) (2) (Fig. 1).

The term microbes means bacteria, algae, fungi, and protozoa. This article focuses on the ecology of bacteria in aquatic environments. A general overview of aquatic microbiology can be found in Ford (3), Hurst et al. (4), or Sorokin (5).

HABITATS

Aquatic systems offer different habitats for microbes to live in: free water, sediment, and hyporheic interstitial (6). Of special importance are interfaces such as the surface of water, sediment, rocks (epilithon), and aquatic plants (epiphyton). Microbes that live in **free water** (pelagic organisms) can be dispersed as single cells or concentrated in aggregates (Fig. 2). Organic aggregates, which are larger than 500 μm , are called lake, river, or marine "snow" (7). They are hot spots of microbial life in aquatic systems and are enriched in nutrients compared to the surrounding water. The formation of aggregates allows direct syntrophic interactions such as the exchange of metabolic intermediates.

Natural waters typically contain between 0.5 and 5×10^6 bacteria per mL (5). Microbial numbers, diversity,

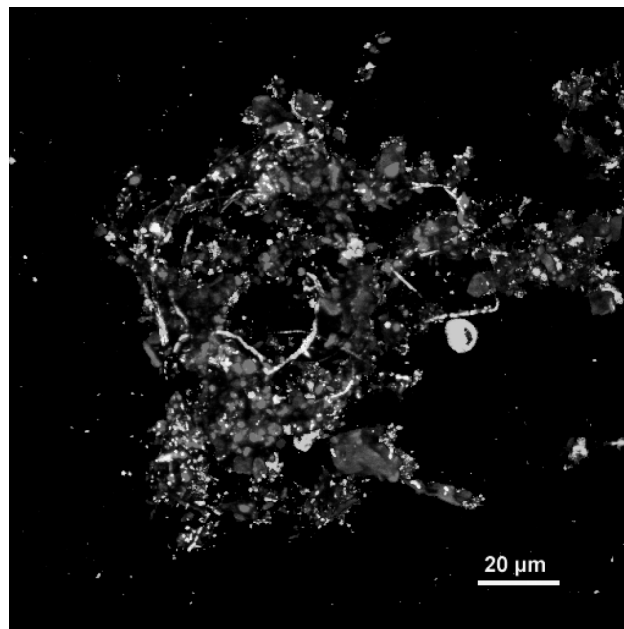


Figure 2. Confocal laser scanning micrograph of a lotic aggregate from the River Saale, Germany. Maximum intensity projection of 31 sections at 1 μm distance from four channels. Color allocation: reflection = white, autofluorescence of algae = blue, nucleic acid staining of bacteria = green, lectin stained glycoconjugates = red (courtesy of Thomas R. Neu) This figure is available in full color online.

and activities are usually higher in the **sediment** and at surfaces than in free water. Sediments typically contain about 10^9 bacteria per mL. The fraction of metabolically active cells is also higher in sediment compared to free water. In a Swedish lake, only 4% of the pelagic bacteria, 37% of epiphytic bacteria, and 46% of sediment bacteria were metabolically active (8). This can have several causes such as higher nutrient availability, higher habitat diversity, or protection against predation. Microbial numbers and activities usually decrease exponentially as sediment depth increases. The absence of turbulent mixing allows the patchiness and spatial (mostly vertical) gradients to form in the sediment. The situation is especially complex in the rhizosphere of aquatic plants.

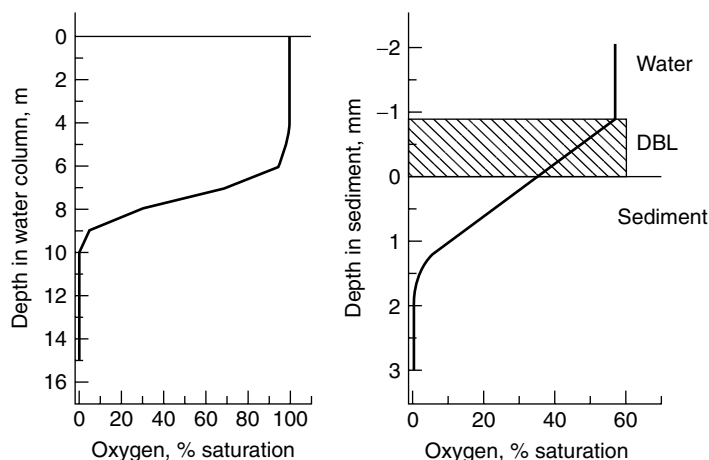


Figure 1. Vertical gradient of oxygen (a) in a lake with anoxic hypolimnion and (b) at the sediment water interface. DBL = diffusive boundary layer.

Surfaces are often covered by biofilms (9,10) or microbial mats (11–13). Living attached to a surface allows microbes to stay in favorable places under conditions of moving water. A biofilm is a “surface accumulation, which is not necessarily uniform in time or space, that comprises cells immobilized at a substratum and frequently embedded in an organic polymer matrix of microbial origin” (10). The solid surface on which a biofilm develops is called the “substratum.” Living conditions in biofilms may differ substantially from conditions in free water. Biofilms are enriched in nutrients, are a place of intensive biotic interactions, and may offer protection against unfavorable conditions. Biofilms are not stable but undergo different successional stages. Biofilm growth starts due to adsorption of cells to a substratum. As the biofilm growth thickens, its composition changes, transport processes influence metabolic activities in the biofilm, and it becomes more and more affected by shear stress which finally leads to detachment and washout of parts of the biofilm.

The liquid–gas interface is characterized by high concentrations of oxygen and hydrophobic substances and high light intensities. It is colonized by a special microflora called neuston (14).

Microbes play an important role in **artificial aquatic systems** such as wastewater treatment plants, biotechnological reactors, and drinking water supplies. Living conditions in such systems can differ substantially from natural habitats and are often characterized by high nutrient content and homogeneity (in reactors) or very low nutrient content (in drinking water). The general habitat characteristics and the adaptations of microorganisms, however, are similar in natural and artificial aquatic systems.

ABIOTIC FACTORS

Water

Per definition, aquatic bacteria live in habitats where water is available. Many natural waterbodies, however, temporarily dry out due to water level fluctuations. Organisms in such habitats develop strategies to survive periods of dessication. They may form spores or other resting stages.

Even if microbes are swimming in liquid water, it might not be available to them because of low water activity. Water activity expresses the ratio of water in the vapor phase to the amount in vapor saturated air (ranging from 0 to 1.0). Low water activity can be caused by high salt concentration. Most bacteria need a water activity higher than 0.98 which is the activity of sea water at 25 °C (14). Especially resistant halobacteria can live at water activities as low as 0.6. They are found in salt lakes or salt saturated evaporation ponds.

Substrates

Microbes need different types of substrates. These include electron donors, electron acceptors, macronutrients (carbon, nitrogen, and sulfur), micronutrients (e.g., different metals and trace elements). Many organisms also depend

on organic substrates. The availability of electron acceptors (O_2 , NO_3^{2-} , Fe^{III} , SO_4^{2-} , CO_2) is a major determinant of which physiological groups of microorganisms are predominant. Surface waters are mostly oxidic, and aerobic microbes dominate. In sediments and under certain circumstances also in free water (groundwater, hypolimnion), conditions are anoxic, and anaerobic organisms dominate microbial populations (Fig. 1). Anoxic bacteria are also frequently found in oxidic environments (15). These can be facultative anaerobes such as denitrifiers and also oxygen resistant anaerobes or strict anaerobes that live in anoxic microsites (e.g., inside microbial aggregates). The oxidic–anoxic interface is a place of high microbial activity and special adaptations (16). At the oxidic–anoxic interface, redox cycling of mobile electron carriers such as Fe^{II}/Fe^{III} or H_2S/SO_4^{2-} can occur on a small spatial scale.

Heterotrophic microbes depend on organic substrates as a carbon source; autotrophs such as photosynthetic organisms or nitrifying bacteria can use CO_2 that is available in most aquatic systems which are buffered by a carbon buffer system. Exceptions are acidic waters where inorganic carbon is available only as physically dissolved CO_2 and potentially limits autotrophic growth.

Concentrations of nutrients are usually low in natural environments. Consequently, cells in natural water are usually much smaller than cells grown in culture media, and generation times are longer. Microbes have developed several strategies to survive starvation. Most obvious is the formation of metabolically inactive resting stages like spores or the increased affinity of nutrient uptake systems. All nutrients have to cross the diffusive boundary layer before they can be taken up by an organism. The diffusion flux to a cell depends on its surface to volume ratio, so smaller organisms are more efficient in diffusive substrate uptake (17). Microbial growth in natural environments is often very slow (18).

Temperature

Temperature is the second most important parameter besides nutrient availability that controls microbial growth in natural aquatic systems. Aquatic microbes can live in liquid water at all temperatures between freezing and boiling. The world record for hyperthermophilic bacteria is currently 113 °C, and many extreme thermophiles cannot grow at “normal” temperatures (19). Such extreme temperatures occur at submarine hydrothermal vents where water remains liquid at temperatures above 100 °C due to the hydrostatic pressure. Most natural aquatic habitats, however, are characterized by low temperatures. In deep lakes and in the oceans, great parts are colder than 5 °C, and microbial processes are slow. The temperature in the natural environment is usually below the optimum growth temperature of bacteria. Even polar or alpine ice is colonized by microorganisms, and the boundary between surface ice and water is a place of microbial life (20).

Light

Microbes are affected by light in different ways, both directly or indirectly. Light is the energy source of

photosynthesis which can take place under aerobic and anaerobic conditions. Anoxic photosynthesis typically takes place at the chemocline where anoxic conditions meet maximum light intensities. High abundances of purple sulfur bacteria can be concentrated in only a few centimeter thick layer of water (21). It is especially important for phototrophic organisms to maintain their vertical position in the water column. To counteract sedimentation, microbes may use active movement or regulate buoyancy by intracellular gas vesicles (22). In moving water, pelagic organisms are continuously transported from regions of high light near the surface to darker conditions further down and back (23).

At the water surface or on mud surfaces, solar radiation can be inhibiting for microbial life. One strategy to prevent photodamage is migration of motile organisms into the sediment during periods of high illumination (24). Microorganisms have pigments to protect them from photodamage. UV radiation can alter nutrient conditions by mobilizing dissolved organic matter (25) or by reducing iron (26).

pH

Natural aquatic systems cover the whole pH range from 0 to 14, and all of them are inhabited by microbes (27). Acidophilic bacteria (28) live in extremely acidic waters. Basophiles and alkaliphiles live in hypersaline waters. The activity of microorganisms can modify the pH of their habitat. The oxidation of iron or sulfur compounds can lower the pH to extreme values below pH 2. Photosynthetic activity can create local pH values higher than pH 10 in phototrophic biofilms (29).

Shear Stress

Water movement affects microbes mechanically in aquatic habitats and is an important regulator of biofilm growth. High current velocities cause detachment and washout of attached microorganisms (9). On the other hand, drift is an import mechanism for colonizing new habitats.

BIOTIC INTERACTIONS

Food Web

Microbes play an important role in the food web in aquatic environments. Bacteria, flagellates, and microzooplankton consume dissolved organic matter (DOM) which is released by phytoplankton. By this **microbial loop** (30), energy and nutrients are returned to the main food chain. In unproductive aquatic systems, bacterial production can exceed phytoplankton production and make these systems net sources of CO₂ (31).

Bacteria **compete** with eukaryotic algae and phagotrophs for resources. Usually, bacteria play a larger role in low productivity (oligotrophic) ecosystems compared to highly productive (eutrophic) systems. Oligotrophic systems that have low nutrient concentrations and higher proportions of soluble organic C favor heterotrophic bacteria to phytoplankton and phagotrophic heterotrophs (32). The microbial mineralization of organic matter is

a complex network of different reactions carried out by different organisms.

Syntrophy

In anoxic environments, the degradation of organic matter resulting in the accumulation of uncompletely oxidized intermediates such as acetate or hydrogen gas is energetically unfavorable. Methanogens and sulfate reducers can, for example, consume hydrogen gas and lower its partial pressure so that the hydrogen producing process becomes thermodynamically favorable. This "interspecies hydrogen transfer" allows the degradation of compounds which are refractory to degradation by the pure culture of a single species (33). A prerequisite of such syntrophic interaction is that the organisms are close spatially.

Grazing

Bacterial numbers in pelagic systems and in surface sediment are often controlled by grazing. Microorganisms have developed different strategies against grazing. They can change cell size, change cell shape to longer forms, or form multicellular aggregates (34). Some bacteria are not digested by higher organisms but live as endosymbionts inside bigger aquatic organisms.

Bioturbation

The activity of macroorganisms can greatly modify the sediment environment (35). Organisms and particles can be moved by bioturbation, and porewater chemistry might be changed. Bioturbation can increase both homogeneity or patchiness of the sediment and introduce short-term changes of conditions.

Bacteriophages

Viruses are the most abundant biological entities in aquatic systems. Typical abundances of these 20 to 200 nm particles are 10¹⁰ per liter in surface water. They probably infect all organisms and influence biogeochemical and ecological processes as well as gene transfer among microorganisms (36).

CONCLUDING REMARK

Due to the development of new methods for studying microbes in their natural environment, we now have a first impression of the complexity of the natural habitat. Many issues such as biotic interactions in aggregates and biofilms, the reaction of microbes to quickly changing environmental conditions, and their distribution in very heterogeneous environments remain important topics for further research. The study of the distribution, role, and function of microbes in their natural aquatic environment is one of the most exciting areas in aquatic ecology.

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CALIBRATION OF HYDRAULIC NETWORK MODELS

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INTRODUCTION

Availability of reliable network modeling software coupled with affordable computing hardware technology has led to rapid growth in the use of hydraulic network models for design, planning, and operational studies of water distribution systems. Friendly graphical user interfaces (GUIs) and integrated computer-aided design (CAD) and geographic information systems (GIS) modeling environments helped drastically reduce model development time and the associated costs. The validity of these models, however, depends largely on the accuracy of input data and the assumptions made in developing the model. Although carefully developed models tend to have greater control on much of the data associated with the model, certain model parameters exist that are either not readily available or

difficult to obtain. Such parameters typically include pipe roughness factors and nodal demands and their spatial and temporal distribution. As a result of the difficulty of obtaining economic and reliable measurements of both of these parameters, final model values are normally determined through the process of model calibration. Model calibration involves adjustment of these and other uncertain network model parameters until the model results closely approximate actual observed conditions as measured from field data. In general, a network model calibration effort should encompass seven basic steps:

1. Identification of intended use of the model
2. Identification of calibration model parameters and their initial estimates
3. Model studies to determine calibration data sources
4. Data collection
5. Macrocalibration
6. Sensitivity analysis
7. Microcalibration

These steps have been successfully tested by the authors on several model calibration studies. Each of these steps is discussed in the following sections.

IDENTIFICATION OF INTENDED USE OF THE MODEL

Before calibrating a hydraulic network model, it is important to first identify its intended use (e.g., pipe sizing for master planning, operational studies, design projects, rehabilitation studies, water quality studies) and the associated type of hydraulic analysis (steady-state versus extended-period). Usually, the type of analysis is directly related to the intended use. For example, water quality and operational studies require an extended-period analysis, whereas some planning or design studies may be performed using a steady-state analysis. In the latter, the model predicts system pressures and flows at an instant in time under a specific set of operating conditions and demands (e.g., average or maximum daily demands). In extended-period analysis, the model predicts system pressures and flows over an extended period (typically 24 hours).

Both the intended use of the model and the associated type of analysis provide some guidance about the type and quality of collected field data and the desired level of agreement between observed and predicted flows and pressures (1). Models for steady-state applications can be calibrated using multiple static flow and pressure observations collected at different times of day under varying operating conditions (e.g., fire hydrant flow test data). On the other hand, models for extended-period applications require field data collected over an extended period (e.g., one to seven days).

In general, a higher level of model calibration is required for water quality analysis or an operational study than for a general planning study. For example, determining ground elevations using a topographic map may be adequate for the latter, whereas the former may require

an actual field survey. Such considerations obviously influence the methods used to collect the necessary model data and the subsequent calibration steps.

IDENTIFICATION OF CALIBRATION MODEL PARAMETERS AND THEIR INITIAL ESTIMATES

The second step in calibrating a hydraulic network model is to determine initial estimates of the primary model parameters. Although most models will have some degree of uncertainty associated with several model parameters, the two model parameters that normally have the greatest degree of uncertainty are the pipe roughness coefficients and the demands to be assigned to each junction node.

Pipe Roughness Values

Initial estimates of pipe roughness values may be obtained using average literature values or directly from field measurements. Various researchers and pipe manufacturers have developed tables that provide estimates of pipe roughness as a function of various pipe characteristics, such as pipe material, pipe diameter, and pipe age. Table 1 shows typical Hazen–William roughness factors (2). Although such tables may be useful for new pipes, their specific applicability to older pipes decreases significantly as the pipes age. In addition, the variation in roughness with pipe age depends on other factors, such as water chemistry, and will vary from location to location. As a result, initial estimates of pipe roughness for all pipes other than relatively new pipes should normally come directly from field testing. Even when new pipes are being used, it is helpful to verify the roughness values in the field because the roughness coefficient used in the model may actually represent a composite of several secondary factors such as fitting losses and system skeletonization.

Adjusting Pipe Roughness vs. Pipe Diameter

The approach of adjusting pipe roughness values assumes that the pressure drop is entirely because of frictional and other minor losses within the pipeline. If the intended use for the calibrated model does not include water quality simulations, this approach of adjusting pipe roughness value will have little or no impact on model results. On the other hand, this approach might seriously affect results from water quality simulations, which is particularly true when a substantial portion of the distribution system comprises fairly old pipelines. Loss of carrying capacity in such pipes is not only because of increased roughness but also because of the reduced flow area resulting from tuberculation, scaling, sedimentation, and biofilm growth. Flow area reductions as high as 50% are not uncommon in certain distribution systems. Although adjusting pipe roughness to reflect the reduction in carrying capacity will not affect hydraulic gradeline and pressure calculations, significant deviations in simulated water quality parameters are possible as the erroneous velocities are used in water quality calculations. Therefore, for more accurate water quality predictions, it might be prudent to adjust the pipe diameters by limiting roughness coefficients to certain predetermined values. For example,

Table 1. Typical Hazen–William Pipe Roughness Factors

Pipe Material	Pipe Age	Pipe Size (Inches)	C Factor
Cast Iron	New	All Sizes	130
		12 and Over	120
		8	119
		4	118
	10 years old	24 and Over	113
		12	111
		4	107
		24 and Over	100
	20 years old	12	96
		4	89
		30 and Over	90
		16	87
	30 years old	4	75
		30 and Over	83
		16	80
		4	64
Welded Steel	50 years old	40 and Over	77
		24	74
		4	55
		4	55
	Values of C the same as for cast-iron pipes, 5 years older		
	Values of C the same as for cast-iron pipes, 10 years older		
	Average value, regardless of age		120
	Large sizes, good workmanship, steel forms		140
Concrete or Concrete Lined	Large sizes, good workmanship, wooden forms		120
	Centrifugally spun		135
	In good condition		110
	Plastic or Drawn Tubing		150

1 inch = 25.4 mm.

if the computed Hazen–William roughness coefficient for a pipeline is 40, one could limit the coefficient to about 90 and compute the pipe diameter that would produce the measured pressure drop.

Spatial and Temporal Distribution of Nodal Demands

The second major parameter determined in calibration analysis is the average (steady-state analysis) or temporally varying (extended-period analysis) demand to be assigned to each junction node. Initial average estimates of nodal demands can be obtained by identifying a region of influence associated with each junction node, identifying the types of demand units in the service area, and multiplying the number of each type by an associated demand factor. Alternatively, the estimate can be obtained by first identifying the area associated with each type of land use in the service area and then multiplying the area of each type by an associated demand factor. In either case, the sum of these products is an estimate of the demand at the junction node. Although in theory the first approach should be more accurate, the latter approach is more expedient. Estimates of unit demand factors are available from water resource handbooks and textbooks (3).

MODEL STUDIES TO DETERMINE CALIBRATION DATA SOURCES

After model parameters have been estimated, the accuracy of the model parameters can be assessed, which is done by executing the computer model using the estimated parameter values and observed boundary conditions and then comparing the model results with the results from actual field observations. Collecting accurate and appropriate field data is crucial for proper calibration of a hydraulic network model. One should use at least 10 to 20 pressure readings measured at locations evenly spread over the water distribution system for steady-state calibration. These pressure readings should ideally be collected when the network is under “stressed” conditions, i.e., during a peak-hour demand condition, during filling cycle of a ground-level pumped storage tank, or during hydrant flow tests. Under these conditions, the increased flowrates make the pressure readings more sensitive to small changes in pipe roughness values. The hydrant flows that produce more than 15 psi drop from static pressures are ideal for calibration studies. In most distribution systems, the peak-hour demands may not provide 15 psi drop at sufficient number of locations. Ground-level pumped storage tanks are not common in

most distribution systems. Therefore, hydrant flow tests provide a logical alternative for pressure reading in most systems. However, not every hydrant flow will result in an adequate pressure drop. Hydrants located at or near large pressure mains, supply sources, and elevated storage tanks tend to provide less than the desired pressure drop. On the other hand, certain hydrants might yield very little flow while showing large pressure drops (e.g., 80 psi of pressure drop while yielding only 200 gpm) because of localized losses such as constricted flow areas. Although a network model that is yet to be calibrated may not be meaningful for planning and operational studies, it can provide some insight into the locations for hydrant flow tests that provide reasonable flows and adequate pressure drops. The following procedure is recommended to locate hydrants for calibration field studies.

- Perform hydrant flow analysis on all hydrants in the distribution system. Hydrant flow analysis appears to be a standard feature in most commercial modeling packages and provides information on flowrate that a hydrant can deliver while maintaining a minimum pressure of 20 psi throughout the distribution system. One could use the nearest junction nodes in place of hydrants if the model does not contain the hydrant information.
- Compute the flowrate per unit pressure drop (difference between static and residual pressure) for each hydrant.
- Rank order the hydrants from lowest to highest of flowrate per unit pressure drop.
- Select about 15 to 25 hydrants from the top of the list that are evenly scattered throughout the distribution system. Care must be taken to avoid hydrants with unusually low flowrates to pressure drop ratios. A closer look at these hydrants and related flows might reveal problems with the network model. The authors have also found that unusually high flowrate to pressure drop ratios could be because of network anomalies that can be corrected by a close observation of network model parameters.

DATA COLLECTION

The next step is to acquire calibration data from field studies. This section describes the general procedures for collecting and recording field data.

Fire Flow Tests

Fire flow tests are useful for collecting both discharge and pressure data for use in calibrating hydraulic network models. Such tests are normally conducted using both a normal pressure gauge (for measuring both static and dynamic heads) and a Pitot gauge (for use in calculating discharge). In performing a fire flow test, at least two separate hydrants are first selected for use in the data collection effort. One hydrant is identified as the pressure or residual hydrant, whereas the remaining hydrant is identified as the flow hydrant. The general steps for

performing a fire flow test are illustrated by McEnroe et al. (4).

In order to obtain sufficient data for an adequate model calibration, it is important that data from several fire flow tests be collected. Before conducting each test, it is also important that the associated system boundary condition data be collected, which includes information on tank levels, pump status, etc. It is a common practice for the local fire departments to conduct hydrant flow tests and record the time of day and corresponding flows and pressures. However, in most cases, such records do not include the boundary conditions associated with each hydrant flow test, as the main purpose for their tests is to rate the fire hydrant and not necessarily for hydraulic calibration. Therefore, care must be taken to avoid hydrant flow data that does not include the associated boundary conditions data. Table 2 may be used as a template when collecting calibration data using hydrant flow tests.

Telemetry Data

In addition to static test data, data collected over an extended period of time (typically 24 hours) can be very useful for use in calibrating network models. The most common type of data will include flowrate data, tank water level data, and pressure data. Depending on the level of instrumentation and telemetry associated with the system, much of the data may be already collected as part of the normal operations. For example, most systems collect and record tank levels and average pump station discharges on an hourly basis. These data are especially useful in verifying the distribution of demands among the various junction nodes. If such data are available, the data should first be checked for accuracy before use in the calibration effort. If such data are not readily available, the modeler may have to install temporary pressure gauges or flowmeters in order to obtain the data. In the absence of flowmeters in lines to tanks, inflow or discharge flowrates can be inferred from incremental readings of the tank level.

In systems with ground-level pumped storage tanks (Fig. 1), the tanks are filled during off-peak demand periods using the high system pressures. Pressures in the distribution system drop dramatically during tank-filling operation, thereby providing an opportunity to use the associated data for model calibration. Tank telemetry data, along with a few static pressure readings within the distribution system where a significant drop occurs, could be used for calibrating the network model in lieu of hydrant flow test data.

EVALUATE MODEL RESULTS

Using fire flow data, the model is used to simulate the discharge from one or more fire hydrants by assigning the observed hydrant flows as nodal demands within the model. The flows and pressures predicted by the model are then compared with the corresponding observed values in an attempt to assess model accuracy. In using telemetry data, the model is used to simulate the variation of tank water levels and system pressures by simulating

Table 2. Sample Hydrant Flow Data Collection Table for Calibration Studies

Test	Date	Time	Flowing Hydrant				Residual Hydrant				Tank Levels			Pump Status			Remarks			
			Name	Elevation	Static Pressure		Discharge Pressure	Flowrate	Name	Elevation	Static Pressure		Residual Pressure	A	B			A	B	
1																				
2																				
3																				
4																				
5																				
6																				
7																				
8																				
9																				
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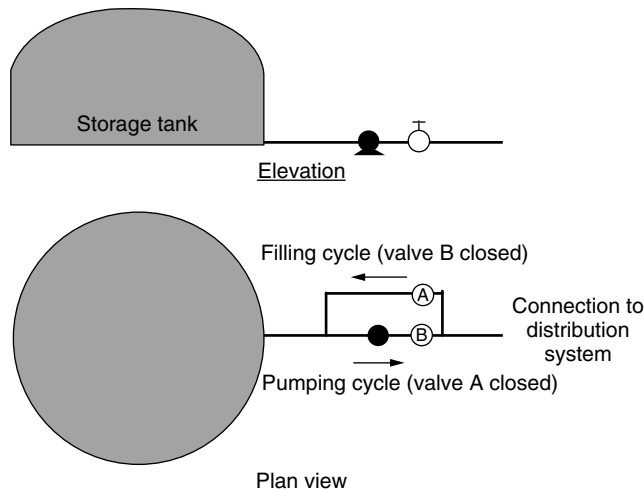


Figure 1. Schematic of a ground-level pumped storage facility.

the operating conditions for the day over which the field data was collected. The predicted tank water levels are then compared with the observed values in an attempt to assess model accuracy.

Model accuracy may be evaluated using various criteria. The most common criteria are absolute pressure difference (normally measured in psi) or relative pressure difference (measured as the ratio of the absolute pressure difference to the average pressure difference across the system). In most cases, a relative pressure difference criteria is normally preferred. For extended period simulations, comparisons are normally made between the predicted and observed tank water levels. To a certain extent, the desired level of model calibration will be related to the intended use of the model. For example, a higher level of model calibration will normally be required for water quality analysis or an operational study as opposed to use of the model in a general planning study. For most applications, a maximum state variable (i.e., pressure grade, water level, flowrate) deviation of less than 5–10% is generally regarded as satisfactory (1).

Deviations between results of the model application and the field observations may be caused by several factors, including: (a) erroneous model parameters (i.e., pipe roughness values and nodal demand distribution), (b) erroneous network data (i.e., pipe diameters, lengths, etc.), (c) incorrect network geometry (i.e., pipes connected to the wrong nodes, etc.), (d) incorrect pressure zone boundary definitions, (e) errors in boundary conditions (i.e., incorrect PRV valve settings, tank water levels, pump curves, etc.), (f) errors in historical operating records (i.e., pumps starting and stopping at incorrect times), (g) measurement equipment errors (i.e., pressure gauges not properly calibrated, etc.), and (h) measurement error (i.e., reading the wrong values from measurement instruments). The last two sources of errors can hopefully be eliminated or at least minimized by developing and implementing a careful data collection effort. Elimination of the remaining errors will frequently require the iterative application of the last three steps of the model calibration process—macrocalibration, sensitivity, and

microcalibration. Each of these steps is described in the following sections.

MACROCALIBRATION

In the event that one or more of the measured variable values are different from the modeled values by an amount that is deemed to be excessive (i.e., greater than 30%), it is likely that the cause for the difference may extend beyond errors in the estimates for either the pipe roughness values or the nodal demands. Possible causes for such differences are many, but may include

- valves that are open in the system but modeled as closed or partly closed
- valves that are closed or partly closed in the system but not modeled correctly
- inaccurate pump curves or pumps defined by useful power rather than head/flow data
- incorrect pump (on/off) status data or pump operating speeds for variable speed pumps
- inaccurate tank telemetry data
- inaccurate flow or pressure gauges
- incorrect pipe sizes and lengths (typographical errors)
- inaccurate pressure regulating valve settings
- incorrect network geometry
- inaccurate junction or hydrant elevations
- hydrants located on wrong pipes near multiple pipe junctions (Fig. 2)
- incorrect pressure zone boundaries
- missing or incorrect demand data—especially pertaining to large users (e.g., hospitals, schools, etc.)

The best way to adequately address such errors is to systematically review the data associated with the model in order to ensure its accuracy (5). In most cases, some data will be less reliable than other data. This observation provides a logical place to start in an attempt to identify the problem. Model sensitivity analysis provides another means of identifying the source of discrepancy. For example, if it is suspected that a valve is closed, this assumption can be modeled by simply closing the line in the model and evaluate the resulting pressures. Potential errors in pump curves can sometimes be minimized by simulating the pumps with negative inflows set equal to observed pumps discharges (6).

Plotting pressure contours pertaining to the simulation that produced undue low pressure (compared with the measured value) might provide an insight into certain problem areas. For example, a significant flow constriction (wrong pipe diameter or partially closed valve) in the model would produce rapid pressure drops that could be easily traced using the pressure contour plots. Probing into headloss/1000 values in the vicinity of the hydrant under question might help trace the problem pipes. In general, it would be relatively easier to trace the problems if the simulated pressure reading is significantly lower than the measured value. Only after the model results

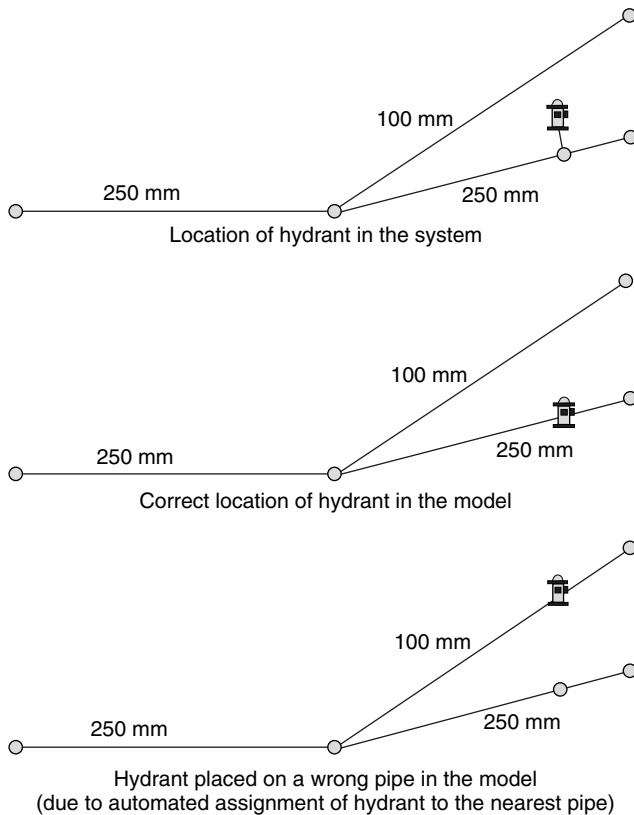


Figure 2. Example depicting misplaced hydrant.

and the observed conditions are within some reasonable degree of correlation (usually less than 20% error) should the final step of micro-level calibration be attempted.

SENSITIVITY ANALYSIS

Before attempting a micro-level calibration, it is helpful to perform a sensitivity analysis of the model in order to help identify the most likely source of model error. This analysis can be accomplished by varying the different model parameters by different amounts and then calculating the associated effect. For example, many current network models have as an analysis option the capability to make multiple simulations in which global adjustment factors can be applied to pipe roughness values or nodal demand values. By examining such results, the user can begin to identify which parameters have the most significant impact on the model results and thereby identify potential parameters for subsequent fine tuning through micro-level calibration.

MICROCALIBRATION

After the model results and the field observations are in reasonable agreement, a micro-level model calibration should be performed. As discussed previously, the two parameters adjusted during this final calibration phase will normally include pipe roughness and nodal demands. In many cases, it may be useful to break the microcalibration into two separate steps: (a) steady-state calibration, and (b) extended-period calibration.

In performing a steady-state calibration, the model parameters are adjusted to match pressures and flowrates associated with static observations. The normal source for such data is from fire flow tests. In an extended-period calibration, the model parameters are adjusted to match time-varying pressures and flows as well as tank water level trajectories. In most cases, the steady-state calibration will be more sensitive to changes in pipe roughness, whereas the extended-period calibration will be more sensitive to changes in the distribution of demands. As a result, one potential calibration strategy would be to first fine tune the pipe roughness parameter values using the results from fire flow tests and then try to fine tune the distribution of demands using the flow/pressure/water level telemetry data.

Historically, most attempts at microcalibration have typically employed an empirical or trial and error approach. Such an approach can prove to be extremely time-consuming and frustrating when dealing with most typical water systems. The level of frustration will, of course, depend somewhat on the expertise of the modeler, the size of the system, and the quantity and quality of the field data. Several commercial software packages offer an optimized calibration feature that would ease some of the frustration associated with model calibration. Genetic algorithm (GA)-based techniques for model calibration appear to be popular with the commercial software packages. The optimized calibration techniques obtain an optimal set of decision variables (e.g., pipe roughness values and nodal demands) that minimize the average deviation between measured and calculated pressures and flows while satisfying all system and bound constraints (7). Although one would expect this to be a one-time process, several factors force repeated use of this approach until a more satisfactory set of decision variables are obtained. One of the reasons for repeated use of optimized calibration runs is the way the pipelines are grouped. GA-based approaches solve for optimal group roughness values instead of roughness values for individual pipelines. Alternatively, they can solve for an optimal multiplication factor for the preassigned roughness values for all the pipes in a group. Pipes of similar roughness characteristics (estimated based on material, age, and diameters of the pipes) are treated as one group. However, this is a highly subjective process and therefore offers opportunities for regrouping and reapplying the optimized calibration technique. Similarly, the explicit bound constraints on the pipe group roughness values offer opportunities for relaxing/tightening of the bounds and reapplying the optimized calibration technique. The iterative procedure for model calibration using optimized calibration approach is summarized in the following.

- Setup the model to include calibration data (observed pressures and flow readings).
- Group the pipes based on similar roughness characteristics.
- Estimate the roughness values for each group of pipes.
- Set up an upper and lower bound for each group of pipes based on estimated group roughness values. As

an initial guess, these bounds should be within 20% to 30% of the estimated values.

- Perform optimized calibration.
- Compare measured and calibrated pressure readings.
- If not satisfactory, or if any of the calibrated group roughness values reach the bounds, relax the bounds by a small percentage and rerun optimized calibration.
- Repeat the trials until satisfactory results are obtained.
- If the bounds had to be relaxed beyond a reasonable value for a group of pipes, it would imply a problem with network data and not necessarily the roughness values. Go back to macrocalibration probing the network in and around the area of concern, and repeat macrocalibration, sensitivity analysis, and microcalibration steps.

Although this procedure outlines the iterative method for obtaining optimal set of group roughness values, it can be easily modified when calibrating the model for other parameters such as nodal demands and their spatial and temporal distribution (7)

SUMMARY AND CONCLUSION

Network model calibration should always be performed before any network analysis planning and design study. A seven-step methodology for network model calibration has been proposed. Historically, one of the most difficult steps in the process has been the final adjustment of pipe roughness values and nodal demands through the process of micro-level calibration. With the advent of recent computer technology, it is now possible to achieve good model calibration with a reasonable level of success. As a result, little justification remains for failing to develop good calibrated network models before conducting network analysis. It is expected that future developments and applications of both GIS and SCADA technology will lead to even more efficient tools.

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NUMERICAL MODELING OF CURRENTS

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INTRODUCTION

Description of Currents

Currents—water movement—appear in all surface waters: rivers, lakes, coastal seas, and oceans. This flow of water is almost always turbulent, which means with many irregular eddies of different size. Currents are sometimes near to steady movement (not changing with time), but mostly the movement is unsteady, significantly changing with time.

In rivers the currents are more aligned nearly in one dimension (1D), although in larger rivers they can be quite complex and three-dimensional (3D). In shallower rivers or coastal seas the modeling of currents is sometimes two-dimensional (2D), but in reality *all* currents are three-dimensional (3D); that is, the water velocities are in all three directions, although often they are an order of magnitude greater in the two horizontal directions than in the vertical direction.

In surface waters the current circulation has different orders of magnitude.

Global currents are large-scale currents, typical for every world ocean. They are often quasisteady in long-term average, and with their enormous transport of heat from the equator to the poles, and vice versa, they significantly affect climate and weather on the continents. The main driving forces are the temperature gradients between the poles and the equator and the wind. Due to the earth's rotation, the main pattern of global currents is typically from the poles toward the equator on the west side of continents (cold currents), and from the equator to the poles on the east side of continents (warm currents). Some of these global currents are the cold Peru (or Humboldt) current along the west coast of South America (toward the equator), the warm Kuroshio current from the equator along the east coast of Asia, and the warm Gulf Stream along the east coast of North America. The last is deflected by the topography of the coastline toward northern Europe, causing a relatively mild climate along the Scandinavian coastline.

Local currents (mesoscale and microscale currents) are currents in smaller regions, as in coastal (or shelf) seas, which are defined as shallower than about 200 m, to distinguish them from the oceans. Of course, local currents at a smaller scale appear also in lakes and rivers.

Importance of Current Circulation

Numerical modeling of currents usually includes modeling of additional parameters that are important in water ecology. When solving practical problems, the following parameters must be determined: (1) water current (circulation), (2) transport of sediments with currents, (3) transport of any contaminants, and (4) biochemical processes of contaminants.

Water current circulation is always the basis for calculation of the transport of any contaminant. But in many cases, knowledge of the current velocities is used directly, for example, in designing river or marine coastal protection, where the computation of current patterns helps determine bank and riverbed erosion and the optimum form of the protection structures. Transport of sediment is also important from two points of view. In river reservoirs, lakes, or coastal bays, sedimentation is often a big problem. Over the years, river reservoirs upstream of dams are filled with sediments and thus lose their useful storage volume. Even more important is the fact that most contaminants are bound to suspended sediments and often over 90% of the contaminant transport is the result of sediment transport with less than 10% in dissolved form. Determining the transport by currents of any contaminants in surface waters is very important. These can be nutrients, affecting the growth of algae and possibly causing eutrophication, or any toxic chemical. Coastal seas often receive large amounts of contaminants from rivers. Many coastal settlements discharge their wastewater (sometimes untreated) into coastal seas. Determination of currents and contaminant transport—most often by numerical modeling—is essential to determine consequences of pollution and to plan the optimum location of wastewater outlets. A special problem is oil spill, where numerical modeling of the spreading oil slick can help in clean up or can help to find the culprit of the pollution. Heat can also be considered as a kind of contaminant. Coastal thermal or nuclear power plants dispose of their waste heat into the sea. As the water temperature should not increase over a certain limit (usually 3 °C over the “natural” temperature), these power plants should be located at sites where strong currents transport away and mix the waste heat. Because contaminants are not only transported and spread (dispersed) over the water body but are also prone to different transformations due to biochemical processes, another quite difficult task of numerical models is to simulate these processes.

The importance of global currents is crucial. As already stated, global currents significantly affect climate and weather on the continents. The El Niño phenomenon is well known to cause catastrophic changes of weather over large parts of continents. Global currents are also important carriers of nutrients and contaminants of all types, and they influence the fish migration. Many oceanographic studies are carried out to determine details of global currents, their changes (e.g., prediction of the effect of future global warming), and their important influence on the environment.

Methods of Determination of Currents

To determine currents in surface waters, there are three methods: numerical modeling, physical modeling, and field measurements. Numerical modeling is the most appropriate methodology. However, most often it should be combined with field measurements and sometimes with physical modeling.

Physical models were widely used before the era of fast computers. Studied regions were built at a reduced scale in large hydraulic laboratories, and water was pumped to the models. Current velocities and other important

parameters were then measured and used to model nature, using physical laws of similitude. However, there are several difficulties in applying this methodology: physical models are expensive and time consuming to build and operate; it is practically impossible to simulate very large regions; and it is almost impossible to reliably simulate biochemical processes.

Field measurements are becoming very important. As numerical models almost always demand calibration and verification of the simulated results, reliable data for comparison can only be obtained by field measurements. This technology is developing very fast. Remote sensing (e.g., satellite images) can provide the pattern of sea surface temperature, or chlorophyll concentration over very large regions. As these patterns depend on transport by currents, this methodology can be used for verification and calibration of numerical models. *In situ* measurements (directly by ships, buoys) are also necessary in combination with the remote sensing technique.

In fact, reliable simulation of currents and contaminant transport can only be attained by the combined application of numerical models and field measurements.

NUMERICAL MODELS

What Is a Numerical Model?

A simplified definition of numerical model is the following: a computer code that can describe or “simulate” a certain physical phenomenon, in this case current circulation and contaminant transport in surface water bodies. The simulation is never perfect, as the real physical phenomenon is too complex, and too many parameters and processes are involved. However, most often, the simulations are accurate enough to essentially help at solving different scientific and engineering problems.

Every numerical model basically consists of three parts: (1) basic equations, describing the physical phenomena; (2) all the necessary input data (e.g., description of topography, data on physical properties of water and contaminants); (3) “forcing factors”—the data about forces that cause water and sediment movement and sources of contamination. The modelers call these “boundary conditions,” because these data determine the behavior of the model and essentially influence the results.

A numerical model can be compared to a physical model: in both cases the form (topography) of the water body must be given (for numerical model in digital form), as well as the physical properties of the water and boundary conditions. The final results are the same in both cases. The result of a numerical model can even be presented as an animation, showing, for example, water circulation—similar to a movie taken on a physical model.

Dimensions of Numerical Models of Currents

One-dimensional models simulate the flow in one dimension only—for example flow along a river. Topography is given with cross sections; the distance between them depends on how big the river is and on the required accuracy. For unsteady flow, the result is time-dependent depth

and velocity in each cross section. Contaminant concentration can also be calculated in every cross-section and at every time step.

Two-dimensional models are usually used in shallow coastal seas or lakes. The computational region (in the horizontal plane) is divided into control volumes of rectangular or other polygonal form (possibly also curvilinear) and bottom and bank topography is given as input data. There is only one control volume from surface to the bottom. The result of simulation is depth and two components of horizontal velocity and possibly contaminant concentration in every control volume and at every time step.

Three-dimensional models are similar to 2D models: the difference is that the computational region is additionally divided into layers along the depth. In this way, the whole computational domain is divided into “bricks” in all three dimensions. Besides the parameters mentioned with 2D models, the vertical velocity component is also computed.

Three-dimensional models are the most sophisticated and basically the most accurate. However, the real accuracy and reliability of the models depend on the accuracy of the input data and on the calibration, verification, and validation of the models.

Forcing Factors

The water movement or, current is caused and influenced by four “forcing factors.”

Wind has an important effect on the water surface. With its shear stress it causes the surface layer of water to move basically in the direction of the blowing wind. Due to the law of continuity, this movement also causes movement of other water layers. If the phenomenon is taking place in a closed basin, layers near the bottom must move in the opposite direction to the wind. In general, the movement is complex and depends on the bottom and bank topography. In larger seas and oceans, the earth's rotation essentially influences the current pattern (1). A good numerical model must take into account all these influences.

Tide influences the current pattern. When simulating currents inside a semiencllosed bay or coastal sea, the boundary condition for the model is represented by the time variation of tidal level at the open boundaries.

Inflow of rivers also affects the currents, with its inflow of mass and momentum.

Density gradients—variations of density with space and/or time—also cause water circulation. The density changes are most often caused by gradients of temperature and salinity; therefore, this type of forcing is also called “thermohaline forcing.”

The relative importance of the four forcing factors can vary with different cases. Wind and river inflow are usually the most important forcing factors in lakes and smaller coastal seas, while tide is the most important factor in small and middle-sized coastal areas. Density gradients and winds are very important for ocean circulation. A good numerical model should include all four forcing factors, although in many cases some less important ones can be neglected entirely.

Numerical Methods

The water movement or current is described by a system of differential equations, basically a scalar continuity equation (mass conservation) and a vector momentum equation, which has as many scalar components as the dimension of the model. Because these equations are nonlinear, the solution is very complex. Basically two groups of numerical methods are used for their solution: finite difference methods and finite element methods. With the first group of methods, the computational domain is divided into control volumes of polygonal forms (possibly with curvilinear sides), while with finite element methods, the domain is divided into different unequal geometric bodies. The finite difference methods are a bit simpler at transforming the differential equations into finite difference forms, while the finite element method can better adjust the control volumes to irregularly shaped boundaries. In the simulation of currents, the finite difference models are more often used.

The number of control volumes into which the computational domain is divided depends on the accuracy of the input data and on the required accuracy of the results. Typically, the domain is divided into 100–150 cells in each horizontal direction, and into 15–30 layers along the depth. Computing time is long and fast computers are needed for the computation of currents.

Verification, Calibration, and Validation of Models

For each simulated case, the applied model has to be checked for reliability.

Verification of a model is the first test, which shows whether the phenomenon is correctly simulated. Then the model should be **calibrated**. Most often there are a few parameters in the model simulations for which the proper values are not accurately known, so a series of simulations are carried out with variable values of these parameters. The simulation results are compared with results of measurements (or possibly results of another model). In this way the proper values of the parameters are determined. (Example parameters are diffusion coefficients for turbulence or dispersion of contaminants.) When the simulated results are in acceptable agreement with measurements, the model is “verified” and “calibrated.” It is then used to simulate other similar cases, for which the model parameters were not calibrated. If the results are in agreement with measurements, the model is “validated.”

CASE STUDIES

A: Transport of Radioactive Contaminants in Japan Sea

The Japan Sea is a large, almost enclosed sea between the Japanese islands, Sakhalin Island, the Siberian coast, and Korea. Its length is about 1800 km, its width 900 km (see Fig. 1), and the maximum depth is about 3900 m. The relatively weak exchange of water with the Pacific Ocean is achieved through four straits (Fig. 1), the widest being Tsushima Strait between Korea and the Japanese islands.

A large amount of radioactive (RA) waste has been deposited in the Japan Sea, off Vladivostok, at a depth

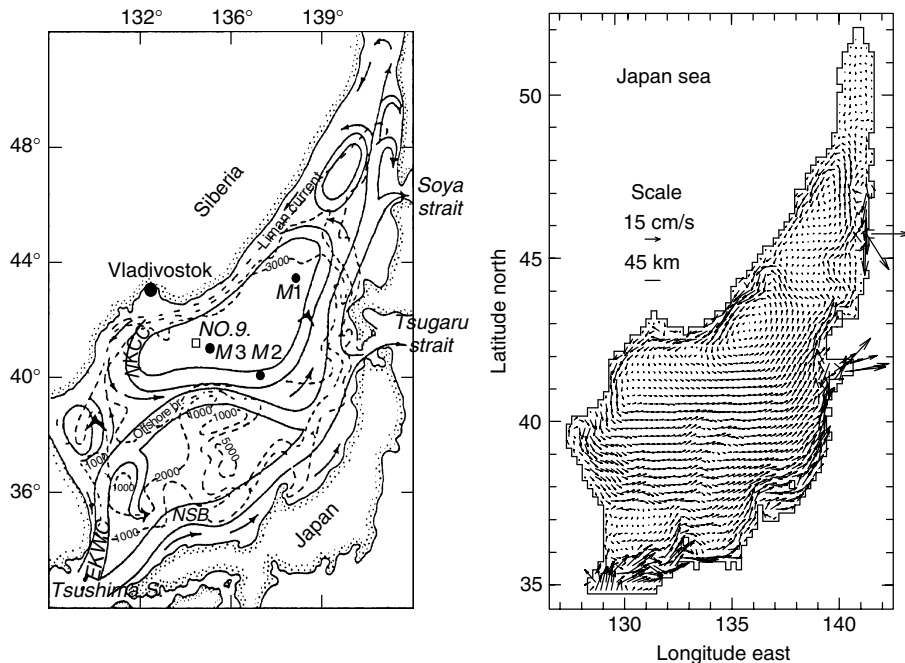


Figure 1. Measured surface currents in the Japan Sea (left) and surface currents simulated by the PCFLOW3D model (right). (From Ref. 3.).

of about 3500 m. Concern existed in Japan that leaking containers with RA waste could cause contamination of fish and other marine organisms in the Japan Sea. A research model should answer the questions: (1) In what concentrations and after what time would the RA contaminants reach the surface layers? (2) Where might they be transported by currents and dispersion? In the simulations, the possible leaking of the radionuclides was supposed to be at location No. 9 (Fig.1).

The first task was to determine the current circulation (4). A 3D numerical model PCFLOW3D was used (2,3). The 3D circulation was driven by three forcing factors: thermohaline forcing, wind, and the inflow/outflow surface currents. The computations were carried out taking into account the measured seasonally averaged temperature and salinity distribution and seasonal winds. A uniform wind field was accounted for (9.5 m/s in winter, 0 m/s in summer). The inflow current through the Tsushima Strait (a branch of the global Kuroshio current) was taken into account.

After some calibration, the results of the model were compared with the observed pattern of the surface currents (Fig.1). It can be seen that agreement is good; the simulated spirals have the same direction and strength as the measured ones. Some measurements also confirmed proper simulation of bottom currents.

On the basis of the computed winter and summer hydrodynamic circulation, transport and dispersion of radioactive contaminants were further simulated. The calculated amount of radioactive contaminants released in 90 d was 1 TBq (10^{12} becquerels), from dumping location No. 9 (Fig. 1).

The final results of the simulations have shown that the RA contaminants would reach the surface mainly in the northern part of the Japan Sea. The first contaminants would reach the surface layers (above 180 m depth, where

fishing is possible) after 3 years; the amount would increase in time and would attain a maximum after about 30 years. But the maximum concentrations would be on the order $1E-2$ Bq/m³, which is far below the natural background values and hence would not represent any danger for the environment.

B: Simulation of Mercury Cycling in the Mediterranean Sea

Mercury is a highly toxic metal, especially in its organic form, monomethylmercury. An international project carried out in 2002–2004 had the goal to determine the natural and anthropogenic sources of mercury in the Mediterranean region and to determine the distribution of the most important mercury forms by the use of a three-dimensional numerical model (5).

Again, the current circulation had to be determined first. It was calculated for four annual seasons. Forcing factors were the following: (1) thermohaline forcing, that is, density gradients due to temperature and salinity distribution (obtained by measurements) in the whole computational domain and also along the depth; (2) wind, and (3) inflow from rivers and exchange of water through the Gibraltar and Bosphorus. Verification and calibration of the model were carried out on the basis of numerous measurements. Figure 2 shows a comparison of simulated and measured currents for the summer season.

Transport and dispersion of mercury were then simulated, sources arriving from rivers, through straits from adjacent seas, and from some known point sources of contamination. Verification and calibration of the transport–dispersion part of the model were carried out on the basis of numerous measurements of relevant parameters. Three cruises by a large research ship *Urania* over the whole Mediterranean Sea provided measurements. Figure 3 shows the distribution of mercury, simulated over one year of inflow from the sources of contamination.

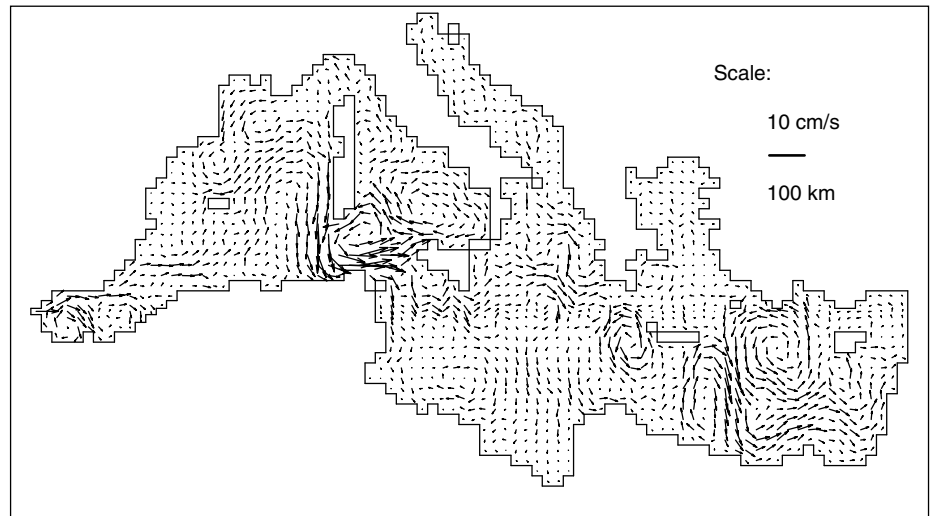
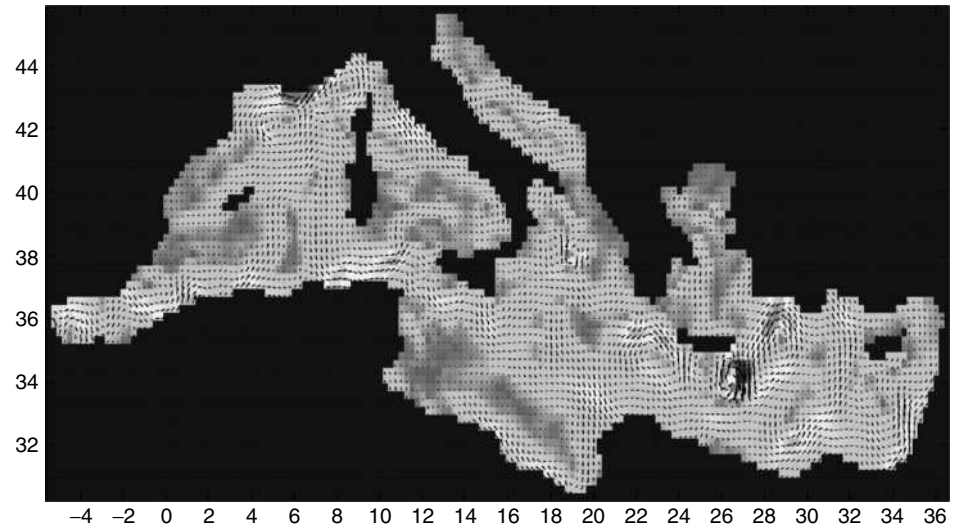


Figure 2. (Top) Measured average currents in the Mediterranean in the summer season. (Bottom). Simulated currents for the same season.

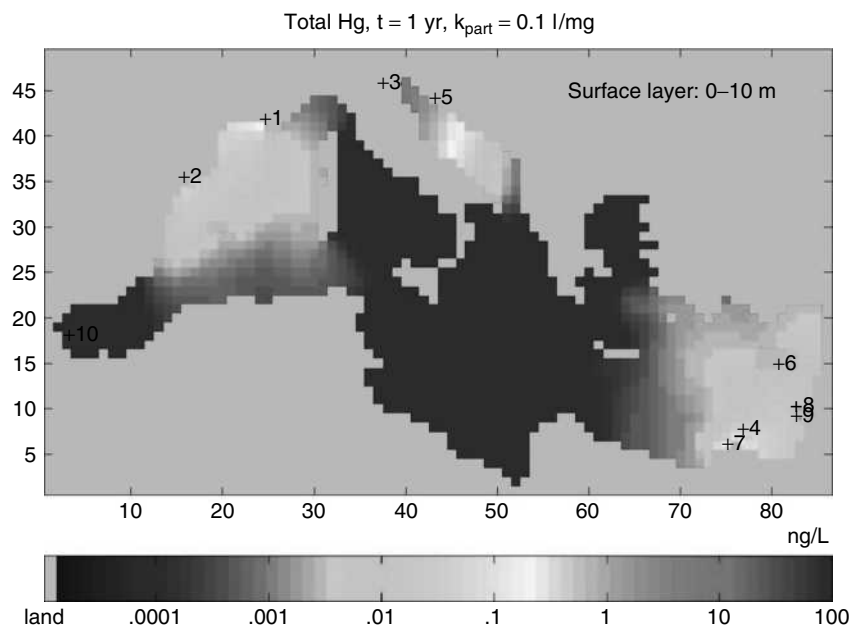


Figure 3. Concentration of total mercury in the surface layer of the Mediterranean Sea, after 1 year of simulation by PCFLOW3D model. Sources of contamination are from rivers and point sources, marked by numbers.

One of the goals of the project was also to couple the water model with the atmospheric model (which also simulates circulation and mercury transport) into a general integrated model. As there is permanent exchange of mercury between the water surface and the atmosphere, the integrated model gives a more realistic result about global mercury cycling and contamination in the region. The general model was finally used as a tool to predict possible measures to diminish concentration of toxic mercury in the Mediterranean region.

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UNCERTAINTY ANALYSIS IN WATERSHED MODELING

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INTRODUCTION

All watershed modeling studies unavoidably contain uncertain elements. This uncertainty is propagated into the model predictions and impacts subsequent decision-making. If not taken into account properly, this can lead to model predictions that are assumed to be more reliable than they actually are. The usefulness of these predictions in any decision-making context therefore becomes questionable.

A watershed model is the numerical representation of our perception of the hydrologic system at hand. It consists of a model structure (i.e., underlying equations), model parameters (i.e., system invariants), and model states. The model is driven by measurements of variables entering the watershed (e.g., precipitation and energy) and compared to observations of the system response (e.g., streamflow or groundwater heads).

SOURCES OF UNCERTAINTY

Sources of uncertainty in this context are manifold and basically include every element that is needed for the

modeling of a watershed as described above. The main uncertainties are listed and described here. The first uncertainty present when modeling watershed systems originates from the modelers' limited understanding of the real world (i.e., uncertainty in the perceptual model used).

Perceptual Model (PM) Uncertainty. A PM (1) is a conceptual representation of the watershed that is subsequently translated into mathematical (numerical) form in the model. The PM is therefore based on our understanding of the real-world watershed system. This understanding might be low, particularly with respect to subsurface system characteristics and water flowpaths, and therefore our PM might be uncertain.

The next group of uncertainties is related to the mathematical/numerical model and its components. These components and the related uncertainties are:

Model Structure (Equation) Uncertainty. The ambiguity present in the PM usually means that a variety of model structures seem to be feasible representations of the watershed at hand prior to any modeling being undertaken. Unfortunately, analysis of the performance of the selected model structures in reproducing observed watershed behavior is often insufficient to distinguish between them and some ambiguity about an appropriate structural system representation remains.

Model Parameter Uncertainty. Models of watershed hydrology necessarily aggregate the natural heterogeneity of watershed characteristics and processes into individual model components. These components are characterized by model parameters that describe, for example, storage size, water residence time, or flowpath. The scale at which these parameters can be measured is commonly different from the scale at which they are applied within the model. At least some of the model parameters therefore have to be estimated through a process of parameter adjustment (manually or automatically), where the parameters are varied until observed and modeled system behavior agree as much as possible. Most watershed models contain a large number of nonlinearly interacting parameters, which makes this a very difficult task, and some uncertainty commonly remains with respect to the question of whether a unique best parameter set has been found.

Model State Uncertainty. With respect to state uncertainty one has to distinguish between initial state uncertainty and state uncertainty after a certain time series of data has been processed by the model. The first uncertainty can usually be taken care of by including a warm-up period that is excluded from the performance analysis. Alternatively, the initial state can be optimized, similarly to the parameter adjustment process described above. State uncertainty during model simulation stems from biases in the model predictions. It can be reduced using data assimilation techniques, that is, approaches that recursively assimilate measurements of the model output variable from previous time steps to adjust the model states to reduce predictive uncertainty.

Variable Uncertainty. This is another, less often considered uncertainty, originating from the problem

of simulated and observed variables not being exactly the same quantity (2). For example, predictions and measurements of soil moisture might be at different scales and therefore a perfect match should not be expected or is not even something the modeler should strive for (3).

Input (Forcing) and System Response Data Uncertainty. There are also uncertainties that are related to the measurements of forcing and response variables. These stem, for example, from an inability to capture the spatial distribution of rainfall or from problems in capturing the true streamflow discharge. These uncertainties can only be reduced outside the modeling process, for example, through better measurement techniques.

Generally, no satisfactory means of considering all these uncertainties simultaneously in a watershed modeling exercise have yet been developed (3). Additionally, the modeler is faced with the problem that, in many cases, the distance between the (erroneous) observation and the uncertain prediction of streamflow is the only measure of deviation (error) at each time step, providing very little information for model evaluation.

APPROACHES TO UNCERTAINTY ANALYSIS

A wide variety of techniques for uncertainty analysis in watershed hydrology are available and often applied (3,4). These methods differ, for example, in underlying philosophy and assumptions and in sampling strategies applied. Little guidance is available to help modelers decide what approach might be the most suitable one for their situation (1). While the differences are large, there is some overlap between all the approaches. In principle, all techniques for uncertainty analysis can be reduced to three basic steps. The main potential approaches that are applied for uncertainty analysis in watershed hydrology are described based on this three-step approach.

1. *Define What Constitutes a Behavioral Model.* The definition of what constitutes a behavioral model can roughly be divided into two types of definitions, those that follow a strict statistical approach, and those using set-theoretic measures not consistent with traditional statistics. The former are based on assumptions about the structure of the model residuals, that is, the differences between observed and simulated system response. Typical assumptions are that the error is Gaussian and has a constant variance in time (3). The second type is less bound by a formal statistical framework, but this is commonly defended with the statement that the structure of the uncertainty is unknown in most modeling studies and that restricting assumptions are therefore not justifiable. Examples of the second type include the generalized likelihood uncertainty estimation (GLUE) (2) framework and fuzzy measures.
2. *Find All Models that Comply With This Definition in the Feasible Parameter Space.* The search for all models that are consistent with the definition consists of two parts. First is the search for all

parameter sets within a certain model structure that provide behavioral predictions. This search is commonly performed using various types of random sampling (e.g., uniform random sampling or types of stratified sampling) or using population evolution based techniques. The use of Monte Carlo Markov chains can be very efficient in this context if the assumptions that have to be made about the shape of the response surface are justified.

The second part of the search is related to the problem that potentially different model structures can provide behavioral predictions for a certain case. This is an even more difficult task since the model space is not just infinite, but also unbounded. No systematic approach exists to explore this search space. Approaches suggested include the use of ensembles of model structure (e.g., in a Bayesian framework) (5) or the inclusion of a more complex error model that takes the imperfection of the selected model structure into account (6).

3. *Propagate the Predictions of All Identified Models into the Output Space While Considering Other Uncertainties (e.g., in the Forcing Data).* Three basic approaches are applied for the propagation stage, first-order, statistical, and set-theoretic. The first approach only propagates the mean and the variance of the predicted variable, based on means and variances of inputs and parameters. An overview of the most popular first-order approaches is provided by Melching (7). Statistical approaches sample from the posterior parameter distributions derived in step 2 and usually make assumptions about the distributions of input uncertainties to derive ensembles of predictions. Ensembles of inputs and/or parameter sets, not necessarily sampled from strict statistical distributions, are also propagated in the set-theoretic approach.

CONCLUSION

Uncertainty is an unavoidable element in any watershed modeling exercise. It has to be considered if the model simulations should be used in a decision-making context. Main uncertainties include those stemming from a lack of understanding of the hydrologic system, the problem of identifying appropriate model structure and parameters, and uncertainties in the measured data. While many approaches exist that can be used for uncertainty analysis, little guidance is available to choose the best approach for a certain study. A three-step structure is common to all approaches and potential implementations are discussed above. Future developments in this area will focus on a more realistic representation of the uncertainties present, techniques that allow for the explicit and simultaneous consideration of all uncertainties, and better sampling strategies.

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WATERSHED MODELING

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INTRODUCTION

Models of watershed hydrology are important tools in hydrological investigations for operational and research purposes. Areas of application are the extension of streamflow time-series in space (1–3) and time (4,5), evaluation of a watershed response to climate (6,9) and/or

land-use variability and change (8,9), calculation of design floods (10,11), as load models linked to water quality investigations (12), real-time flood forecasting (13–15), and the provision of boundary conditions for atmospheric circulation models (16,17). This multitude of, often very specific, tasks and circumstances has contributed to the development of a vast number of watershed models starting in the early 1960s (see Todini (18), for a historical review). These models are usually a mixture of linear and nonlinear functions, combined to represent those processes occurring in the specific watershed and important for the study objectives at hand. Watershed models are also used in water quality studies, but the focus here is simulation of the quantitative response and general features of these models.

WATERSHED PROCESSES

The hydrologic cycle is a closed system in the sense that no water is lost. Only part of the processes that constitute the hydrologic cycle are relevant for watershed modeling. A watershed is therefore an open system with respect to inputs (mainly precipitation and energy) and outputs (mainly streamflow, but also surface and subsurface losses). A schematic description of the main processes in a typical watershed system is shown in Fig. 1. Models of watershed hydrology represent these processes using different mathematical approaches and different levels of spatial detail. It is important to consider that the part of the watershed system reproduced in the model usually depends on the prevailing processes and also on the modeling purpose and available data (3). For example,

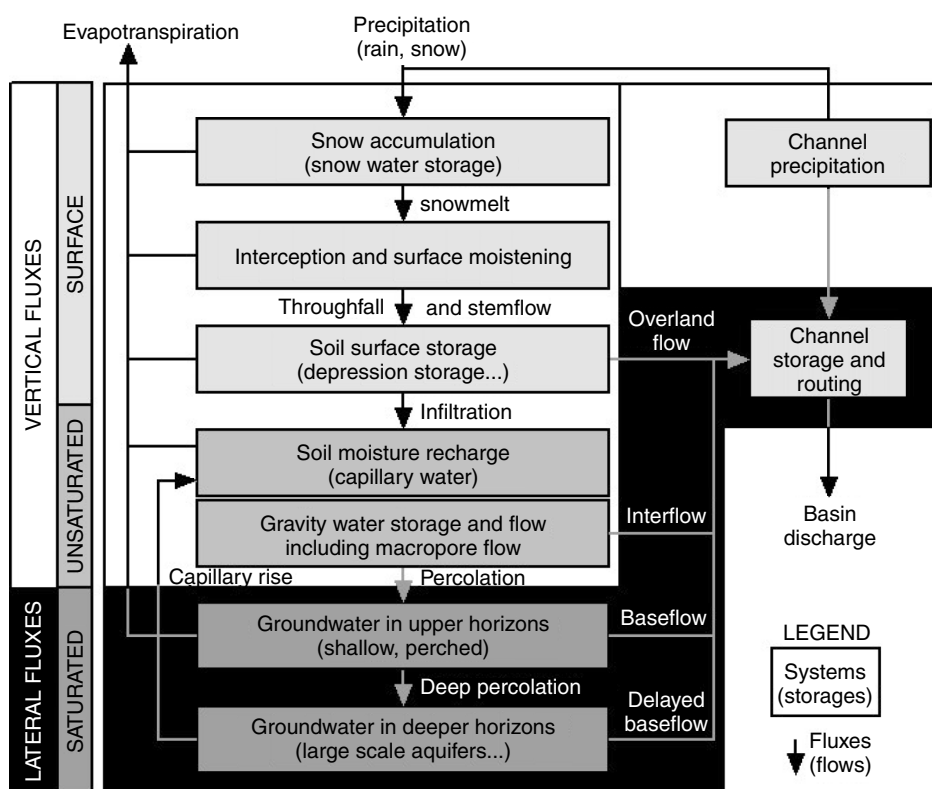


Figure 1. Schematic of watershed processes (after Reference 19).

routing processes are usually not required if the modeling purpose is simulation of the watershed outflow at a monthly time-step.

WATERSHED MODEL CLASSIFICATION

It is common to classify watershed models into three distinct types (20; Fig. 2): (1) *empirical* (also called data-based, metric, or black-box), (2) *conceptual* (also called parametric, explicit soil moisture accounting, or gray-box) and (3) *physically based* (also called mechanistic or white-box) model structures. There are, however, models that have characteristics of more than one group.

Empirical models use available time series of input and output variables (precipitation, streamflow, temperature, etc.) to derive both the model structure and the corresponding parameter values. They are therefore purely based on the information retrieved from the data and generally do not include prior knowledge about catchment behavior and flow processes, hence, the name black box. Popular examples of empirical models are artificial neural networks (ANN; 21,22) and transfer functions (TF; 23). Empirical models are usually spatially lumped; they treat the catchment as a single unit. Variants on purely empirical TF models are *data-based mechanistic models*. They constrain the degrees of freedom to those structures that are physically interpretable, thereby using the hydrologist's understanding of the natural system (24–26).

Conceptual models (3,27) are built from storage elements that represent parts of the watershed where water is temporarily stored, such as soil, aquifers, or streams. These elements are filled through fluxes such as rainfall, infiltration, or percolation, and empty through evapotranspiration, runoff, drainage, etc. Based on a conceptualization of the real-world watershed, the modeler, specifies the structure of these models a priori, for example, the number of and connections between storage elements. These models are also usually spatially lumped. This means

that large parts of the (heterogeneous) watershed are integrated in a single (homogeneous) element. The consequence of this process is that the model parameters lose some of their physical meaning and cannot be measured on the required scale. Rather, the modeler has to use observations of the watershed response to find appropriate values for the model parameters (see next section). Some models are based on a semidistributed approach, in which different models represent individual subwatersheds. This approach is finding increased attention in the literature (28). Conceptual models form the large majority of models used in operational hydrology. Their dependence on measurements of the watershed response (mainly streamflow) limits their use in ungauged watersheds. However, research is ongoing to resolve this problem (3,29).

Physical models (30,31) are based on conservation of mass, momentum, and energy. The spatial discretization applied is usually based on grids, but sometimes also on some type of hydrologic response unit or triangular irregular networks (32). Physical models are therefore particularly attractive for studies that require a high level of spatial detail, such as studies on soil erosion or diffuse-source pollution (33). The initial idea underlying these models was that the degree of physical realism on which these models are based would be sufficient to relate their parameters, such as soil moisture characteristics and unsaturated zone hydraulic conductivity functions for subsurface flow or friction coefficients for surface flow, to physical characteristics of the catchment (18), thus eliminating the need for observed system response to condition the parameters of the model. However, the currently available physical models do not fulfill this ideal. They suffer from extreme data demand, scale-related problems (e.g., the measurement scales are different from the process and model (parameter) scales), and overparameterization (34). A number of key parameters—applied to a large number of elements—still has to be estimated from measurements to capture the uniqueness of a specific watershed (35). The expectation that these models could be applied to ungauged catchments has therefore not been fulfilled (36). Beven (34) argues that this type of model is applied in a way similar to lumped conceptual models, though on a different scale.

A general characteristic of all current models is that they do not provide reliable predictions in ungauged basins (3). The general behavior of a watershed is reasonably constrained by the model structure (i.e., the constituent equations), but some adjustment of model parameters is required to capture the unique characteristics of the system under investigation. A large number of popular models of watershed hydrology can be found in Singh (37) and Singh and Frevert (38,39).

WATERSHED MODELING PROCEDURES

Many authors have suggested procedures for using watershed models. Examples are given in Beven (32) and in Anderson and Burt (40). They vary slightly in the steps included and definitions used. A procedure using commonly found elements is shown in Fig. 3 (after

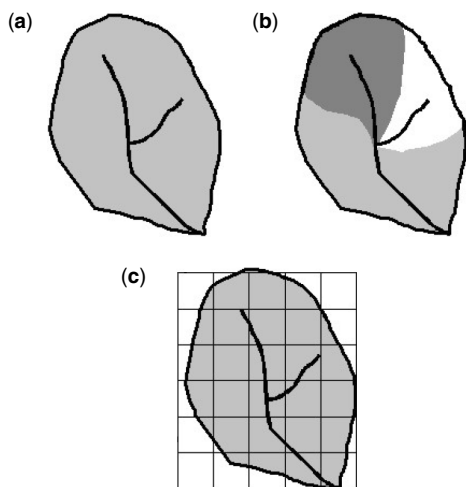


Figure 2. Classification of watershed models according to spatial discretization used: (a) lumped, (b) semidistributed, and (c) distributed.

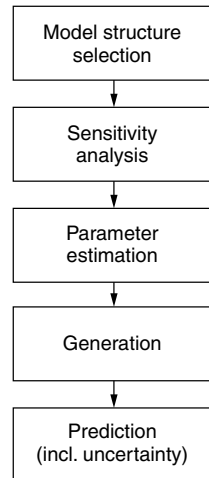


Figure 3. Typical elements of a watershed modeling procedure.

Reference 3). The elements considered are model structure selection, sensitivity analysis, parameter estimation, generalization, and prediction (including uncertainty estimation).

The first step for the modeler is to *select* (or develop) a *model structure*, a combination of mathematical equations, that describes the watershed system as a function of the system characteristics (e.g., saturation or infiltration excess dominated runoff production), the modeling objective (e.g., do the effects of routing have to be considered if only the monthly response is wanted?), and the available data (e.g., a simple evapotranspiration function has to be used if measurements of variables such as humidity or wind speed are not available). Personal preferences and experience in using a particular model structure might also play an important role in the selection process. Woolhiser and Brakensiek (41, p. 15) conclude that “*objective methods of choosing the best model (structure) have not yet been developed, so this choice remains part of the art of hydrological modelling.*” However, future modeling tools might include

knowledge-based approaches to support the decision-making process of the hydrologist for selecting an appropriate model structure.

Sensitivity analysis tests the sensitivity of the model output to changes in the model parameters (or even to uncertainty in the model input or in other model elements). This can be done before parameter estimation to find parameters that are particularly important for model performance—the estimation process can then be focused on these, or it can be done after the calibration as part of the uncertainty analysis—to identify parameters that are particularly uncertain. The simplest approach is a perturbation technique where one parameter is varied over its feasible range while the remaining parameters are kept at reasonable values, although more sophisticated approaches exist (32).

As stated before, the current generation of watershed models generally requires some sort of *parameter estimation* to enable the model to match observed and calculated system behavior, for example, to give reliable predictions. This process is sometimes referred to as calibration. Within this step, the performance of selected parameter sets in reproducing the observed system behavior is evaluated (Fig. 4), either by visual inspection or by calculating some function F that summarizes how well simulated and observed system behavior match:

$$F = f(e) \quad (1)$$

where f is some aggregation function applied to e , the vector of residuals calculated as

$$e_t = z_t - y_t \quad (2)$$

where z is the observed system response, for example, a time series of streamflow; y is the calculated system response; and t is the time step. The modeler attempts to make the function F as small (or large depending on the chosen formulation) as possible, either by manually adjusting the parameters or through some automatic procedure. See Duan et al. (42) for a state-of-the-art review of current techniques.

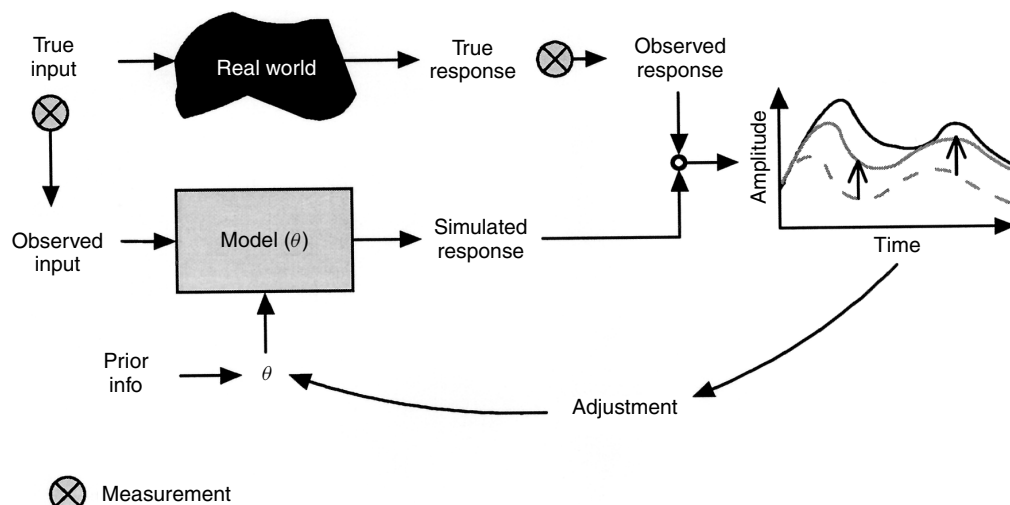


Figure 4. Calibration of a watershed model. The symbol θ represents a parameter vector, y is the system output (usually streamflow), and t denotes time.

The problem with this step of conditioning the parameters on observations is that it can result in parameter values that depend on the specific time series used (e.g., a particular climate period). Some modelers include a *generalization* stage to avoid this. During this stage, the model (including the parameter set found most representative of the system at hand) is applied to another time series of measured system response, not used during parameter estimation, but usually of the same variable. A model is assumed generally applicable if its performance does not degrade significantly during this process (43).

The model can then be used for *prediction*. However, there is a wide variety of factors that introduce *uncertainty* into the modeling process. These are for example, the inability of the modeler to find a “best” parameter set, uncertainties in measuring input and output data, and problems in finding an appropriate model structure to represent the system under investigation. These uncertainties have to be considered and incorporated in the model predictions. It is therefore sensible to state these uncertainties explicitly and include confidence limits that denote the ranges in which the predictions of the model are most likely to fall (44,45).

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MODELING OF WATER QUALITY IN SEWERS

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Modeling of water quality in sewers emerged during the late 1990s. The water quality and amount of water in sewers are dominant factors for designing wastewater treatment plants and for analyzing water quality impacts

from sewer overflows. In addition, sewage under anaerobic conditions may cause corrosion of pipes, malodors, and pose health risks for sewage workers. Hence, it is of utmost importance to analyze and understand the transport and transformation of wastewater in sewers. Ideally, modeling of sewer systems and wastewater treatment plants should be based on the same general principles and done in integrated models to pursue an integrated understanding and management of sewers, wastewater treatment plants, and receiving waters. However, it must be stressed that the transformation of wastewater in sewers is not identical to the processes in activated sludge and biofilm systems—so even though several similarities exist between sewers and wastewater treatment plants, modeling of water quality in sewers is a discipline of its own. This article focuses on how to model water quality in sewers.

THE SEWER AS A PHYSICAL, BIOLOGICAL AND CHEMICAL REACTOR¹

To model water quality in sewers, it is necessary to know the various types of sewers and their water quality characteristics. For a detailed description of the processes in sewer, refer to specialized literature (1). Around the world, all sewers can basically be divided into three categories:

1. storm sewers
2. wastewater sewers
3. combined sewers.

Storm Sewers. Storm sewers dominantly transport rain water, mainly runoff, during rain from city areas. Typically, storm water runoff has low concentrations of pollution and the concentration time for the storm sewer system is usually rather short—of the order of minutes to a few hours. If the concentration time is short and the concentrations are low, then aerobic conditions will dominate, and there will be only a low level of transformation and biological activity in the storm water.

Sanitary Sewers. Sanitary sewers transport wastewater from residences, commercial areas, and industries. Depending on the local layout of the sewer system, the wastewater may be transported by gravity or in pumped lines (in areas with insufficient ground slope available). The concentration time for sanitary sewers may be up to 1–2 days. Wastewater typically has high concentrations of pollution and biodegradable material, so wastewater sewers are potentially a place of high biological activity.

Combined Sewers. Combined sewers transport just wastewater in dry weather and both wastewater and

¹The terminology “The sewer as a physical, chemical and biological reactor” has been largely promoted through the work of the Sewer Systems & Processes Working Group under IWA and in particular promoted by Prof. Thorkild-Hvitved-Jacobsen, Aalborg University, Denmark, and Prof. Richard Ashley, University of Bradford, UK.

runoff from city areas during rain. The concentration time in a combined sewer system, may be rather, short or it may rise to the order of 1–2 days. During dry weather flow, combined sewers operate like sanitary sewers, but during rain, rainwater that has a lower concentration of pollutants is added, and the transport time of the water to the wastewater treatment plant is significantly reduced. When a combined sewer system is overloaded, the excess water is discharged into receiving waters (a river, lake, or a coastal area), and this combined sewer overflow (CSO) may seriously deteriorate the water quality in the receiving waters.

The impact on recipients from overflows from sewer systems can be divided into three typical problems:

1. toxic impact due to un-ionized ammonia discharged mainly from CSOs
2. oxygen depletion
3. hygienic problems caused by fecal coliform.

Hence, the selection of an appropriate modeling approach for water quality in sewers depends both on the type of sewer system and the nature of the problem. Un-ionized ammonia in a combined sewer system may be accurately modeled by applying an advective dispersion model, considering that ammonia is conservative. However, it is a must to include biological and chemical processes to model hydrogen sulfide in a pumped wastewater pipe whose retention time is more than 3–4 hours. The following sections focus on modeling various types of water quality problems in sewers.

DETERMINISTIC MODELING OF WATER QUALITY

The strength of deterministic models for water quality in sewers is increased understanding of the individual processes and their interaction. Deterministic modeling of water quality in sewers and urban drainage can be divided into two groups:

1. quasi-conservative pollution
2. nonconservative pollution.

Modeling water quality in sewers is a challenging task, which assumes that the model is built and verified step by step. Note that a small error in the hydrodynamic model is carried 100% into the water quality model. A volume error of 10% more water in the hydrodynamic model will directly imply that the concentration of pollution in the water quality model will be too small. At present, modeling of pollutant transport may be calculated by using an advective–dispersion model such as MOUSE TRAP (3). Models transporting the pollution by pure advection (no dispersion) are InfoWorks (4) and the SWMM models (EPA SWMM, MIKE SWMM, and XP SWMM) (5). As software for modeling sewers develops rapidly, it is suggested that the reader always seek the latest information from the software suppliers of the models.

Modeling of Quasi-Conservative Pollution

The term *Quasi-Conservative Pollution* covers pollution that is not significantly transformed while it is in the sewer system. Ammonia and phosphorous are water quality parameters, which certainly are not 100% conservative, but which in most cases can be modeled with sufficient accuracy assuming that they are conservative (6,7).

Dissolved conservative pollution can typically be modeled by assigning a measured concentration to the wastewater and to the rain. The resulting concentration in the sewers is then effectively a dilution of the wastewater based on the land use and local industrial load to the sewer system. At this point, it is important that the model applied for pollution modeling represents the transport of pollution inside each pipe. The model must have computational grid points within each pipe otherwise there will be too much dispersion in long pipes, resulting in lowered peaks of concentration.

Modeling of Nonconservative Pollution

The term *Nonconservative Pollution* covers situations where biological and chemical processes play a significant role during the time the water passes through the sewer/drainage system. In this case, modeling the pollution by an advective–dispersion model will be insufficient and will give misleading results. Some examples of *Nonconservative Pollution* are BOD, COD, and hydrogen sulfide. If the sewer system has a very short concentration time, even BOD and COD may be modeled by ignoring the water quality processes, but a contribution from erosion of sediment deposits in the sewers must still be included.

The advective–dispersion model basically has only one calibration parameter, the dispersion coefficient, because advection of the pollution depends fully on the hydrodynamics. To determine the dispersion coefficient, a tracer measurement can be done with salt, rhodamine, uranine, or a similar substance, or the model can be calibrated based on measurements during rainfall of a quasi-conservative pollutant such as ammonia. A tracer study is highly recommended because it verifies that the model accurately describes the transport of a conservative substance under known flow conditions. In addition to finding the dispersion coefficient, the tracer measurement program serves three purposes:

- verification of the assumptions about infiltration to the sewer system
- the possibility of fine-tuning the Manning numbers along the sewers in the tracer measurement
- validation of the capability of the modeling system to describe the physical transport of the dissolved pollutants.

In conclusion, a tracer study serves well as a verification process of a hydrodynamic model. If the model fails to reproduce the tracer results, then there is no reason to continue the study to model anything related to water quality. The final set of input data to the advection–dispersion model consists of

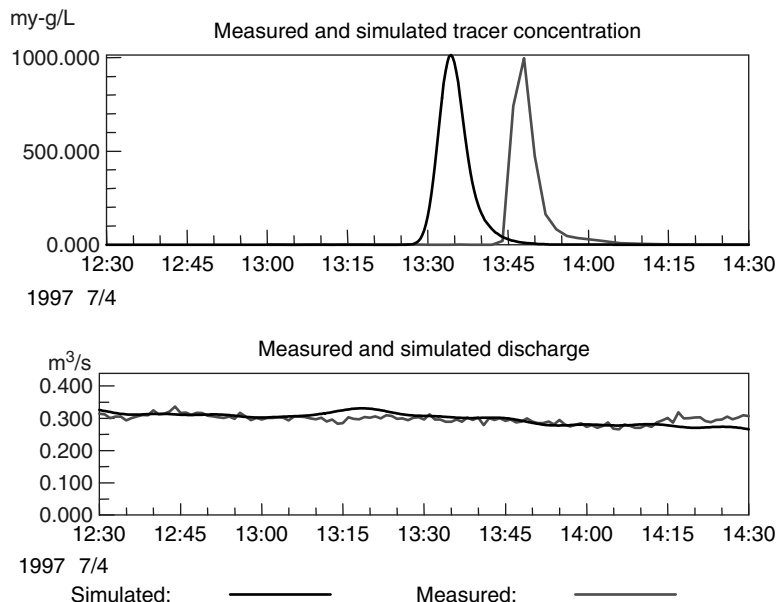


Figure 1. Example of tracer study results, showing that the hydrodynamic model needs to be recalibrated.

1. diurnal variation of dry weather flow
2. diurnal variation of concentration
3. infiltration
4. rainfall runoff
5. major industrial loads in terms of flow and concentration.

The discrepancies between a tracer study and advective dispersion, shown in Fig. 1, are for the city of Helsingborg, Sweden (7).

During the measurements, the tracer dose was added as a half-minute long pulse. From the figure, it can be seen that there was a very good match between the measured and simulated flow, but the timing of the measured and the simulated peaks was out of phase for the tracers. It was found that this occurred because the Manning numbers were never calibrated properly and only a flow gauge and not a water level gauge was available at this location in the sewers. Simulation results by applying an advective dispersion model for modeling various water quality parameters can be seen in Fig. 2 for the Helsingborg sewer system in Sweden.

Sediment Attached Pollution

Pollution may be attached to sediments, especially BOD, COD and phosphorous (2,8). When suspended sediment or sediment deposits exist in sewers, it is of utmost importance to include a model that can also simulate sediment transport processes in the sewers—including erosion and deposition of sediments. Principles for modeling sediment transport in sewers can be found in Mark et al. (9) and IWA (2).

Modeling Nonconservative Pollution by Application of a Water Quality Model

Nonconservative pollution in sewers must be modeled by describing the water quality processes along with

the advective dispersion mode, preferably based on a hydrodynamic model. At this time, only the MOUSE TRAP model has such capabilities (3). The SWMM models (EPA SWMM, MIKE SWMM and XP SWMM) and InfoWorks (4) have only an advection model.

The transformation of pollution may either be modeled by applying a model based on decay rates or by using a model based on the physical, biological, and chemical processes, such as the WATS model (1). The WATS model has thus been successfully applied to wastewater modeling in the Estoril sewer system in Portugal (10). Good case studies demonstrating the applicability of water quality process models during rain are still lacking at this time.

To calibrate a water quality process model, it is of utmost importance to have a minimum of two stations measuring hydraulic and water quality parameters in the important sewers in the catchment because the water quality parameters will determine a process rate, which can be found only by comparing the measurements from two or more measurement stations. Some typical water quality processes under aerobic conditions can be seen in Fig. 3.

MODELING HYDROGEN SULFIDE

Understanding and modeling hydrogen sulfide is important because it introduces problems such as corrosion of pipes and other metal constructions in sewers. It is toxic, it poses a serious health risk to the sewer maintenance staff, and it releases noxious odors to the city environment. Further, it may inhibit sewage treatment processes (11). A high sulfide content may be toxic to fish in streams affected by sewer overflow (12). Hydrogen sulfide develops under anaerobic conditions in sewers, and it interacts with anaerobic water quality processes (1,13). Hydrogen sulfide has traditionally been modeled by using empirical equations such as: Thistlethwayte (14), Boon and Lister (15), Pommery & Parkhurst (16), and Nielsen

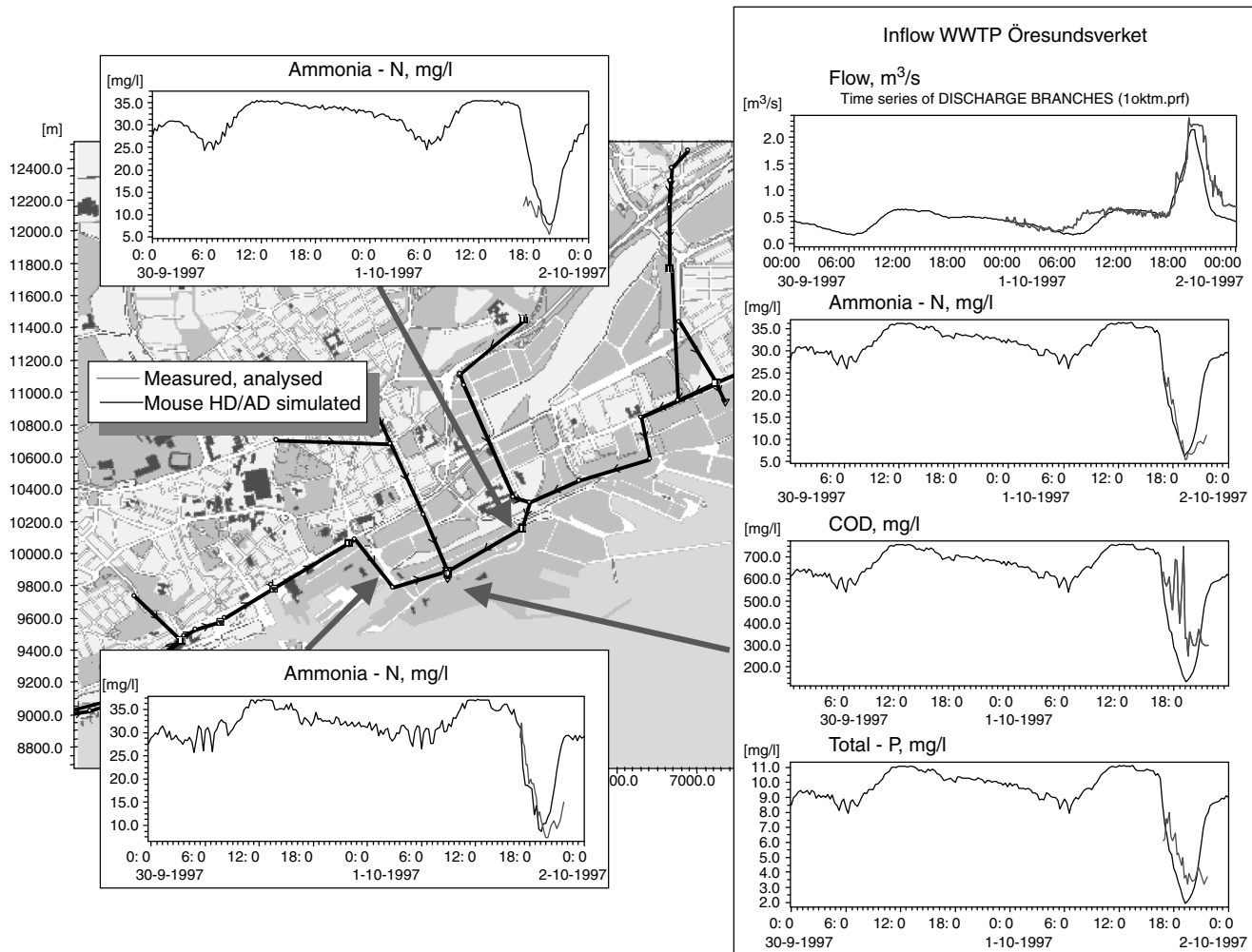


Figure 2. Wet weather sampling. Examples of measured flow and wastewater analyses compared to preliminary MOUSE hydrodynamic and advective dispersion simulation results.

and Hvitved-Jacobsen (17). Note that the four empirical models in these references were developed by using data from pressure mains, so they are not generally applicable to all sewer systems. At present, the only hydrogen sulfide model in commercial software is the MOUSE TRAP model (18). This model includes an advective dispersion model and four empirical equations for sulfide buildup. In addition, oxygen demand and extra Michaelis–Menten expressions are included in the model, so it approaches a more deterministic description of sulfide production.

A more recent and generally applicable model for hydrogen sulfide in sewers is the WATS model (1). The WATS model applies an integrated aerobic and anaerobic concept to the transformation of organic matter and sulfur in wastewater (1).

MODELING FECAL COLIFORM IN SEWERS

Experience modeling fecal coliforms has shown that coliform cannot be considered conservative or without natural decay in the receiving waters (20), for example, in the sea, where the main controlling parameters

for bacterial decay are salinity, temperature, and light intensity. At present, limited knowledge is available concerning the transport of coliform in sewers; it is modeling fecal coliform in sewers by using a first-order decay rate is suggested (21); it is conservative when the concentration time in the sewer system is short—up to a few hours (22).

A PROCEDURE FOR MODELING WATER QUALITY IN SEWERS

In most cases, sediment affects pollution significantly, and when the concentration time of the sewer system is long (as a rule of thumb, 3–4 hours), the interaction of BOD/COD and oxygen must be taken into account in the model to produce reliable results. When the oxygen level plays a significant role, for instance, in modeling hydrogen sulfide, a water quality process model is a must. A consistent procedure for water quality modeling is outlined here. Each step in the modeling procedure must be completed and verified before continuing to the next step. If the model results at a step are unsatisfactory, then it is not feasible to continue because the outcome of each step depends on

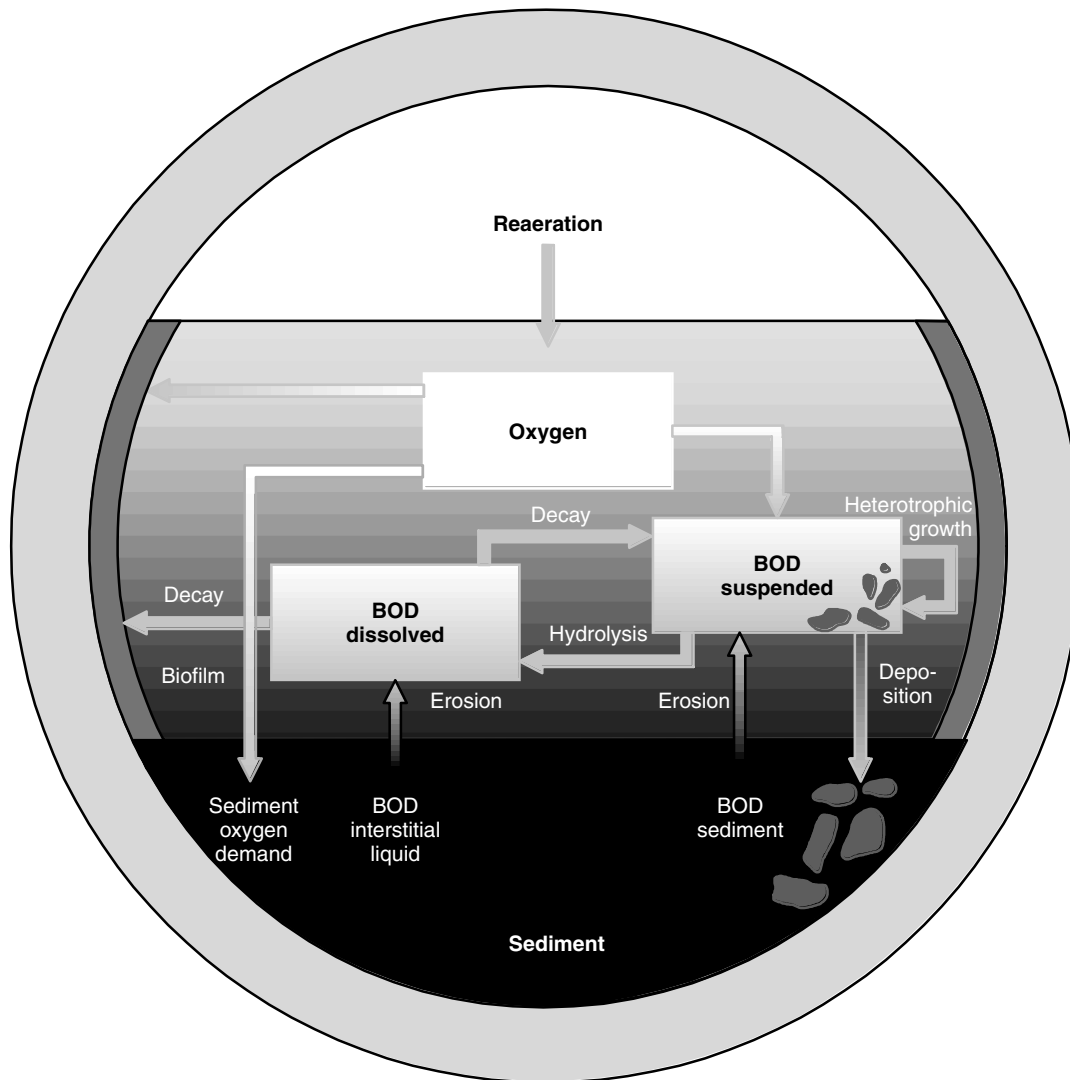


Figure 3. Example of water quality processes in wastewater under aerobic conditions.

the success of the previous steps. The modeling procedure is as follows:

1. Make a tracer verification of the hydrodynamic model, and apply a pure advection and/or advective–dispersion model.
2. Model a real quasi-conservative pollutant, for example, ammonia for both dry weather flow and rainstorms.
3. Analyze the ratio of dissolved and particulate COD and BOD, and check if a “first flush” exists.
4. If the water quality parameters in the study are attached to sediment, then model the sediment transport of the suspended solids.
5. To determine the water quality process rates, obtain measurements of flow and pollution for relevant periods at upstream and a downstream locations in the sewer system.
6. Model dissolved COD and BOD using the process rates from step 5.

7. Model total COD and BOD as the outcome of the modeling of the dissolved COD/BOD and the sediment transport of particulate COD and BOD.

LONG-TERM SIMULATION OF POLLUTION

High computational speed has made it feasible to carry out hydrodynamic and water quality simulations applying deterministic models. These models allow computing annual loads of pollutants to a WWTP or annual loads discharged from CSOs. In addition, extreme events can be modeled, such as those giving rise to high CSO discharges of water and pollutants that can cause fish kills. Through long term simulation, extreme events can be assigned return periods, providing information concerning how often various events occur. In Fig.4, the result from a 3-month simulation is seen. Long-term simulations generally require data spanning three to five times the (return) period of interest.

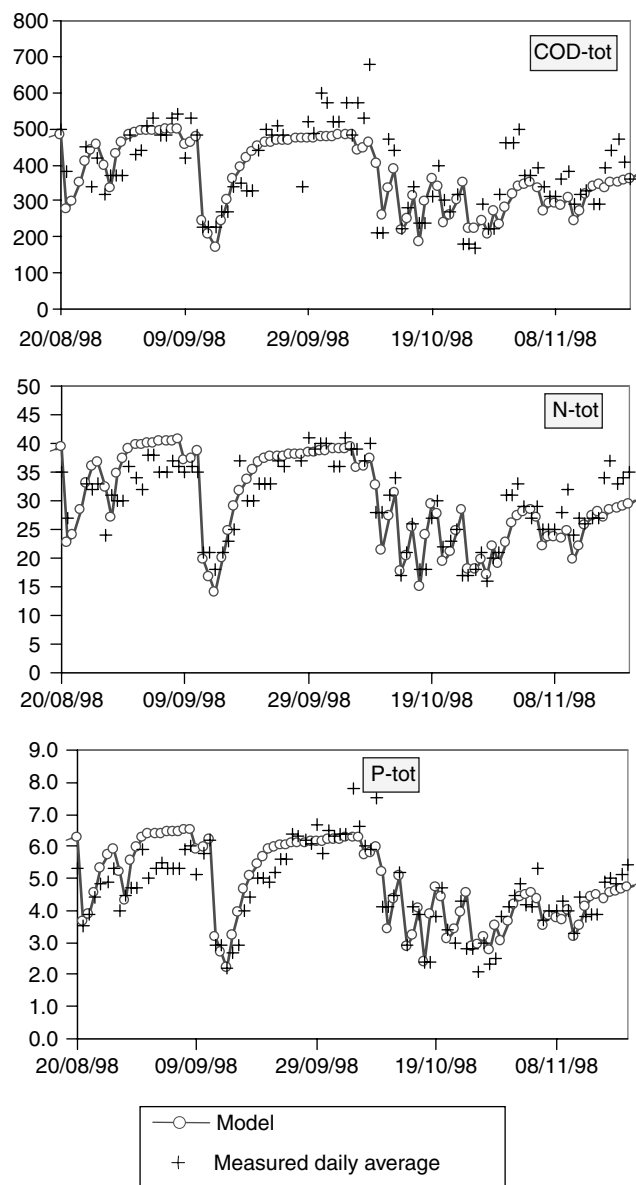


Figure 4. Simulated ammonium concentration in the inflow to the Øresundsverket, WWTP, Helsingborg, compared with analyzed 24 hourly values (each workday).

FUTURE PERSPECTIVES AND CHALLENGES IN RELATION TO MODELING OF WATER QUALITY IN SEWERS

At present, it is feasible to model water quality and pollution in sewers. However, for successful modeling, it is of great importance to understand the hydrodynamics, the transport of pollution, and the potential associated biological and chemical processes. Further, a model describing the dominant processes is essential. And, finally, a systematic procedure must be applied whereby the performance of the water quality model is verified step by step.

Holistic analysis of the sewer system, the wastewater treatment plant, and receiving waters is still an emerging area that has potential for integrated analyses of overall performance. The sewer system and the water quality

processes in the sewers play a key role in this analysis, and, as more information becomes available concerning modeling of water quality in sewers, more solutions, and hopefully, better and sustainable solutions may be developed for handling the water in sewer systems.

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MODELING OF URBAN DRAINAGE AND STORMWATER

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INTRODUCTION

The construction and operation of urban drainage and stormwater systems have historically been driven by the objectives of maintaining public hygiene and preventing flooding. All over the world, cities have grown rapidly during the last century, and city authorities have built and continuously expanded storm and wastewater drainage facilities. Many old cities have developed according to their varying historical needs and visions. Hence, the layout and design of the infrastructure have gradually developed into rather complex systems, which require tools to handle such situations.

An understanding of the physical system and its interaction with the environment is a prerequisite for effective planning and management of urban drainage and stormwater systems. In this respect, computer models provide the opportunity for well-structured analyses of the current performance of a system and the effect of potential changes in the system; they offer a sound scientific framework for coordinated management and planning. Apart from assessing different scenarios, the models also help to improve process understanding. Urban drainage models are used to understand the often rather complex interaction between rainfall and overflow/flooding. Once the existing conditions have been analyzed and understood, alleviating schemes can be evaluated, and the optimal scheme implemented.

Today's advances in computer technology allow many cities all over the world to report overflows from their sewers yearly using a combination of measurements and modeling, and in many places, they manage local and minor flooding problems using computer-based solutions. This involves developing computer models of the drainage/sewer system, for instance, by using software such as InfoWorks (1), MIKE URBAN (2), MOUSE (3), PCSWMM (4), and SWMM (4). All are hydrodynamic modeling packages that solve the one-dimensional equations of St. Venant.

TYPES OF MODELS USED FOR URBAN DRAINAGE AND STORMWATER SYSTEMS

Any model is a simplified mathematical representation of a physical system. This representation may be based on a **deterministic** approach (i.e., a fixed relationship between physical disturbance and its effect, such as model results) or it could be **stochastic**, involving terms of probability in the model inputs and the interpretation of model results. Models for urban drainage and stormwater are predominantly deterministic models, and the text below refers only to this class.

Furthermore, deterministic models applied to urban drainage can be roughly classified as **physical models** and **conceptual models**. Placing a model in one of these classes relies on the level of mathematical sophistication of the treatment of underlying physical processes in the model. Practically, models are classified according to the importance of empirical parameters for the models' ability to describe these processes accurately. The reliance on empirical parameters classifies the model as conceptual and emphasizes the need for validation against field measurements. In this context, various hydrologic models would belong to the class of conceptual models; a hydrodynamic network model is an example of a physical deterministic model.

BRIEF HISTORY OF URBAN DRAINAGE MODELING

Modeling of urban sewer and drainage systems dates back to the late 1960s when the U.S. EPA funded the development of the first version of the EPA Stormwater Management Model (SWMM). The first version 1.0 of SWMM was released in 1971. Around this time, Wallingford (UK) and DHI Water and Environment (DK) developed their first one-dimensional flow models, but it was not until 1982–86, with the release of the commercial software packages, MicroWASSP (1982), MOUSE (3), and PCSWMM (4), that they were no longer 'labeled' as 'in-house software.' These modeling packages came very soon after the first PCs hit the market. Since the release of personal computers (PC), modeling software for urban drainage has closely followed and fully used the full powers of the PC. As PCs have become more powerful, the modeling software has increased in complexity, and the size of the urban drainage model has gone from a maximum of around 250–500 pipes in the mid-1980s to modeling the drainage systems of mega cities, (e.g.,

models with thousands of pipes). Computer modeling of urban drainage has not added much new within the basic theory of urban hydrology and hydraulics, but it brought fast and user-friendly computations of the St. Venant equations to the broad engineering community. During the 1990s, modeling of urban drainage broke new ground by developing both theoretical and modeling approaches for sediment transport and water quality in sewers, processes which today are still not fully understood. Lately, major modeling packages have been fully embedded in Geographical Information Systems (GIS) to ease model building based on the registrations in the databases of the local authorities and to present model output in conjunction with the cities' infrastructural data.

CALIBRATION AND VERIFICATION OF MODELS: FIELD MEASUREMENTS

Urban drainage models are expected to reproduce the behavior of the modeled system with a high level of accuracy. This is usually ensured through parameter calibrations and the verification of model results against the measured performance of the real system. Calibration involves adjusting the model's key parameters, to minimize the differences between the model results and the field measurements (e.g., water levels). A continuous period or a set of intermittent events used for calibration should include a full range of expected operational conditions in the system. Verification provides proof that the model generates results within the acceptable error range. Reliable verification should be carried out for a simulation period or intermittent event(s), which are different and independent of those used for the model calibration.

All models must, if possible, be calibrated before application. Conceptual models require more attention in this respect compared to fully deterministic, physical models because the assumptions of the conceptual models must also be proven valid. Thus, in urban drainage modeling, the focus of model calibration is usually on hydrologic models (conceptual), whereas the deterministic hydrodynamic models of the drainage network typically require only minor adjustments for accurate performance. However, independently of the applied model type, good modeling practice requires thorough model verification before application.

Reliable field measurements in sewers and drains and the results of laboratory analysis (in the case of water quality modeling) are essential for successful model verification (i.e., for the model application as a whole). The harsh environment in sewers and urban drains (poisonous gases and pathogenic microorganisms in confined conduits, aggressive fluids, floating debris), as well as particular hydraulic conditions (alternating free-surface and pressurized flow, rapid flow surges, etc.) requires special monitoring equipment and highly trained specialists for good results. Routinely measured variables include water level, flow velocity, conductivity, temperature, pH, etc. The chemical and biological properties of the water, as well as sediment characteristics,

are obtained by manual or automated sampling, followed by laboratory analyses.

APPLICATIONS OF MODELS TO URBAN DRAINAGE AND STORMWATER SYSTEMS

Urban drainage and stormwater models are usually set up and applied for specific analyses, and a model of the same part of a city may be differently schematized and calibrated for different objectives, based on the application type. The next sections briefly describe common applications of urban drainage and stormwater modeling:

- modeling of water quality
- modeling of urban flooding
- modeling the interaction between urban drainage systems, receiving waters, and wastewater treatment plants.

MODELING OF WATER QUALITY

Modeling of wastewater is important for predicting the inflow of both water quantity and quality to wastewater treatment plants and for precise descriptions of overflow from sewer systems (overflow of a mix of wastewater and rainwater from combined sewer systems)—combined sewer overflows (CSO) and overflow from sanitary sewer systems (SSO). Traditionally, overflows are modeled by using a rainfall runoff model as input to a hydrodynamic pipe flow model in conjunction with a diurnal description of the wastewater produced by the people living in the city.

In the past, the performance of a sewer system was characterized by most Environmental Protection Agencies (EPAs) by the number of overflow events per year. If a sewer system had less than, e.g., five overflows/year, it was considered satisfactory performance. This was a performance criterion which was feasible to check and implement, but it could not account for the actual volumes spilled. It was realized that five overflows/year could be spilling one thimble of water five times, or it could be huge volumes of water spilled five times. When modeling was introduced, it was possible to compute volumes from all overflows in a drainage system. In the past, they all had to be measured to get a picture of the complete overflow from the system. Relatively cheap and reliable application of models for calculating overflow volumes and pollution emissions for the entire system changed the EPA focus from a frequency/year to a consideration of the yearly overflow volumes, whenever that was more appropriate. For instance, for lakes, the accumulated overflow of nitrogen and phosphorus from the drainage system is most important, whereas the number of spills is less important. For smaller streams, the loads from extreme events causing high ammonia or low oxygen concentrations may be the dominant processes. The approach to sewer rehabilitation/optimization was changed from a reduction of the number of spills/year to a reduction of the yearly overflow volumes together with consideration of the load to receiving waters. In other words, "The focus went downstream."

The load to streams and rivers would typically be computed based on a conceptual hydraulic model such as the SAMBA model (5). This kind of model is simplified so that it includes only computation of a time–area runoff hydrograph together with the most significant elements in the pipe/drainage system, weirs, pumps, locations with divergent flow, and outlets. Due to the simplifications, the calculations are very fast and suitable for scanning a large number of alternative layouts of drainage systems. In the early days of computer modeling, this kind of modeling was the only way to calculate annual loads and extreme statistics of discharges of water and pollutants based on historical time series of rain events. The model focused on the requirements from the EPA.

The introduction of water quality modeling of urban drainage such as MOSQUITO (6) and MOUSE TRAP (7) in the middle of the 1990s, extended the possibility of hydrodynamic modeling of “clean” water to include modeling of the transport of pollution and sediments in the sewer and drainage systems. This further enhanced the description of the modeling of wastewater and the pollution loads to the receiving water because this kind of water quality modeling provides an accurate description of the dilution process when rainwater mixes with sewage, together with an estimate of the water quality processes in the system. The outcome was more accurate computations of concentrations of conservative pollutants in the CSO and, for example, ammonia can be modeled accurately in most cases. For more information concerning modeling water quality, refer to the section in the Encyclopedia of Water, **Modeling of Water Quality in Sewers**.

During the 1990s, long-term simulations of drainage network hydraulics and water quality gradually became standard practice, facilitating simulation of long historical time series of rain and their impact in terms of overflow or load on the wastewater treatment plant. This combination of the simulation of wet weather conditions by using the dynamic pipe flow model and a simple hydrologic model to simulate the dry weather conditions makes it possible to simulate many years (>20 years, but feasible length of simulation period depends on model size, computer speed, and rainfall data duration) continuously within a reasonable computational time on a PC. The results can be presented based on the statistical occurrence of impacts, for example, in terms of overflow volume. Understanding the behavior of the system and its interaction with the environment, gained from long-term simulation, is superior to simulation of selected single rains because the uncertainty from selecting a few representative rains is removed. Furthermore, based on a long-term simulation, the statistics are made on the computed output (the impact of the rain) in terms of CSO, SSO, or flooding. In the past, the statistics were made during the selection of the rains because as they would be input for the simulation.

MODELING OF URBAN FLOODING

The problems arising from urban flooding range from minor ones, such as water getting into the basements of a few houses, to major incidents where large parts of cities are inundated for several days (Fig. 1). Most modern



Figure 1.

cities in developed countries typically have smaller scale problems, due to locally insufficient sewer capacity. Other cities, for instance, in Asia-Pacific, can have more severe problems because there is insufficient drainage and much heavier local rainfall. The situation is further aggravated because cities in developing countries grow rapidly these days, but do not have the funds necessary to extend and rehabilitate their existing drainage systems. For more information concerning urban flooding, refer to the section in the Encyclopedia of Water, **Urban Flooding**.

Modeling urban flooding differs significantly from modeling pipe flow, which has a confined geometry can be measured accurately and easily. The flow is basically one-dimensional and accurately described by the St. Venant equations. When the water leaves the drainage/stormwater system, it flows onto a three-dimensional ground surface where the flow routes and paths are not well defined. The surface must be described by a digital elevation model containing roads and houses in the flooded areas. Modeling surface flooding takes on additional data in an environment where the local geometry is difficult to describe accurately and the flow pattern may be one- or two-dimensional. Recently, a model was developed that combined a one-dimensional pipe flow model and a two-dimensional surface flood model (8). This approach allows exchanging surcharge water and flow in the pipe network. The results can be used to identify hazardous flood zones, where high velocity occurs (9). Another two-dimensional model using a shallow water equation has been developed to study the flow characteristic of street flooding when water escapes from manholes (10). Modeling urban flooding is an emerging area where applications have started to appear during the last decade. For further information, refer to Mark et al. (11).

Examples of cities that experience urban flooding problems on a larger scale are Bangkok, Thailand; Dhaka, Bangladesh; Ho Chi Min City, Vietnam; Mumbai, India; and Phnom Penh, Cambodia. In the year 2000, nearly 17,000 telephone lines in Mumbai went out of service after flooding occurred, and the electricity supply was switched

off for a long time in most areas for safety reasons. The floods reached 1.5 m deep in the locations that were most severely inundated. Fifteen lives were lost in the Mumbai flooding (Times of India, July 25, 2000). In Dhaka, even a little rain can inundate parts of the city for several days. The situation was highlighted in September 1996, when the residents of Dhaka experienced ankle to knee-deep water in the streets. Heavy traffic jams occurred because of the stagnant water on the streets, and daily activities in parts of the city were almost paralyzed. In 1983, Bangkok was flood inundated for six months, which caused direct damage worth approximately 6,600 million baht (\$149 million) (12).

MODELING THE INTERACTIONS AMONG URBAN DRAINAGE SYSTEMS, RECEIVING WATERS, AND WASTEWATER TREATMENT PLANTS

"Integrated modeling" in urban drainage is a term describing *"Modeling of the interaction between urban drainage systems, the receiving waters, and wastewater treatment plants."* The term, "receiving waters" is used here for groundwater, streams, rivers, lakes, the sea, etc. Most urban areas have these three components. Few who are involved in managing these systems would disagree with the statement: "It is logical and beneficial to plan, design and operate these components in an integrated manner" (13).

Integrated modeling is a complex exercise due to the sheer size of the model problem and also due to the different modeling approaches for each subsystem that has developed during modeling history. The hydrodynamics in flow and water levels can be directly transferred between subsystems, but presently water quality modeling is handled completely differently in channels/rivers and at the wastewater treatment plant. This causes problems in the interface and data transfer between subsystems, which again impacts accurate modeling of WQ processes.

The logic behind integrated management is old, and a number of separate tools exist. The advent of computer models has been the biggest single factor in the development of integrated analyses. Because of technology limitations in the past, historical reality has been removed from integration, and independent pragmatic criteria for each component have been the norm (13). These are some examples.

- Sewers are typically designed to contain storms of a specific return frequency without surcharging, rather than protecting against flooding.
- Combined sewer overflows are engineered to pass forward flow dictated by downstream sewer or WWTP capacity rather than the ability of the receiving water to accept the overflows.
- Treatment plants themselves are often designed to cope with an arbitrary multiple of dry weather flow, rather than treating all flow from the sewer system.

In most parts of the world, there have been fragmented planning and management of the sewer system, the wastewater treatment plant, and receiving waters.

Furthermore, there is a general lack of methodology and technology available for integrated planning and management. Finally, there is simply a lack of a regulatory framework for "integrated thinking." An exception to this "rule" is the UPM procedure (14) in the United Kingdom, which is highly recommended reading. Today's challenge is to move the legislation from individual consideration of each subsystem's performance to integrated management of the urban wastewater system.

The quality of an integrated model for an urban drainage system depends strongly on the quality of the WQ sewer model, which today is the weakest link in integrated modeling. Despite these uncertainties, integrated modeling is still very beneficial, and the future will surely be dominated by integrated modeling.

REAL-TIME MODELING

At present, hydrologists working in urban areas are facing many new challenges imposed by the ever-changing hydrologic environment in cities. In such conditions, emphasis is focused on managing the urban systems as efficiently as possible within the existing infrastructure by applying currently available information and technology, where real-time modeling plays an increasingly important role. It occurs in two types of applications:

- modeling real-time control and active real-time control of urban drainage systems
- modeling as a real-time decision support tool.

MODELING REAL-TIME CONTROL AND ACTIVE REAL-TIME CONTROL OF URBAN DRAINAGE SYSTEMS

In the early 1990s real-time control of sewers (RTC) was recognized as a big leap forward to achieving superior operation of many sewer systems. Since then, RTC has certainly been proven an eminent solution for some sewer systems to reduce the risk of flooding and the amount of CSO at the same time. But RTC has not been implemented to the extent it was foreseen in the early 1990s, and it seems as if RTC never became "the big thing" that was envisaged. Some of the reasons behind the slow implementation of RTC in the real world are due to the risks of applying RTC, and the safety precautions which have to be taken to ensure smooth and secure operation of the RTC system in case of a power failure or other abnormal incidents.

The aim of RTC modeling of an urban drainage/stormwater system is to develop the strategies applied in real life for pumps, weirs, and gates and to evaluate the impact and risk from malfunctions, in a system controlled by RTC. For example, what is the flood risk if a weir or a gate is stuck during a rain?

Today RTC is applicable for

- reducing the flooding risk by using the storage capacity within the sewer system,
- reducing pollution spills from sewer overflows by retaining more stormwater and sewage within the

sewer system as a result of attenuated flows and storage,

- reducing capital costs by minimizing the storage and flow carrying capacity requirements within the system,
- reducing operating costs by optimizing pumping costs and providing the information necessary to implement effective maintenance procedures, and
- enhancing wastewater treatment plant performance by balancing inflow loads and allowing the plant to operate closer to its design capacity, thereby reducing the variability of the final effluent.

When the feasibility of RTC is evaluated for a sewer system, it is important to be aware of the changes in flow pattern introduced by RTC. If RTC is applied during dry weather flow, for example, to level out the flow to a wastewater treatment plant, it must be investigated if the new flow conditions introduced by using RTC cause sediment deposits to develop in new and undesired places. If this is the case, then sediment deposits may increase the risk of flooding, or the entire saving gained by applying RTC may be spent on increased cleaning and maintenance. Neither would be a good outcome.

MODELING AS A REAL-TIME DECISION SUPPORT TOOL

Applying a model in conjunction with real-time data (e.g., rain data) provides means for improving the information about the current status of the urban drainage system. In general, rain gauges provide information only at specific locations. The application of a model provides the possibility of filling information gaps between the rain gauges and computing the future impact of the rain. Models can be automatically executed in real time, for example, when the rainfall at specified locations exceeds preset threshold values. Based on the forecast by a hydrologic model or an urban drainage model, the level of information about the hydrologic system can be extended into the future. Real-time hydrologic information makes it possible to detect potential hazards shortly before or just after they happen, for example, heavy rainfall recorded in certain parts of the catchment that causes flooding, or overflows, and actions can be taken to mitigate the problems in the drainage system. Another example is that local traffic control can redirect traffic to roads not flooded or provide decision support to the operators of pumps and weirs in the system (15).

The application of real-time hydrologic information is not the full solution to overflow and flooding problems, but it may mitigate flood problems and in this way reduce the economic losses and the stress on people and the environment. Finally, the information generated by a real-time hydrologic information system can be applied by using the historical data for design and maintenance analyses to achieve better functionality of the urban hydrologic system before the next heavy rain arrives. In this fashion, the system can also be used to train operators.

FUTURE PERSPECTIVES AND CHALLENGES IN MODELING URBAN DRAINAGE

Calibration Support for Urban Drainage and Stormwater Systems

Today, urban drainage and stormwater models are calibrated to reproduce measurements of flow, water levels, and velocity through a manual calibration procedure. The reliability of the modeling is highly dependent on the adequacy of the calibration procedure. Traditionally, the model is calibrated manually using a trial-and-error parameter adjustment procedure. At present, however, automatic calibration support is being widely discussed (16) as a replacement for the manual calibration procedure, which is subjective and can be time-consuming. Automatic methods for model calibration take advantage of the speed and power of computers, and are objective and relatively easy to implement. Automatic calibration supports are still emerging. So they need to be checked, and the calibration must be thoroughly evaluated by an experienced engineer because the automatic calibration procedure may stop at a local minimum of the objective function, or it may not converge sufficiently toward the chosen objective due to deficiencies in the measured data.

WATER QUALITY MODELING

Modeling water quality in sewers is still an emerging area because water quality and sediment transport processes are difficult to describe precisely and the theory is still developing. At present, a number of practical operational problems in sewers push the research toward sustainable urban drainage systems and practical solutions to handle significant problems such as hydrogen sulfide production. Hydrogen sulfide causes a noxious smell in the public environment; it corrodes pipes and pumps in sewer networks; and poses a health risk to people working inside sewers, for example, during maintenance. When a sewer pipe collapses due to corrosion from hydrogen sulfide, it imposes huge costs on society to restore the pipe. Therefore, there is a need to understand and manage the phenomenon of hydrogen sulfide formation, and the emerging development of modeling tools for this problem makes it possible to assess causes and alleviating measures.

MODELING THE INTERACTION BETWEEN URBAN DRAINAGE SYSTEMS, RECEIVING WATERS, AND WASTEWATER TREATMENT PLANTS

The future challenge within integrated modeling is to develop a consistent modeling formulation for water quality in sewers, the wastewater treatment plant, and receiving waters, which can be carried across the three subsystems. Furthermore, water quality processes in sewers are at present still an emerging area of research, and need more research. Finally, legislation must be revised nearly all over the world to deal with permits to systems which are optimized from a holistic point of view. Today, around the world, most legislation focuses

only on one single sub-system, independently of the rest. This will open optimization of the total urban drainage system rather than optimizing each subsystem individually.

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MODELING ungauged watersheds

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INTRODUCTION

Models of watershed hydrology are irreplaceable tools in today's research and practice. Their areas of application are wide reaching, from water management and flood forecasting, to their use as load models for water quality studies. A vast number of these models have been developed since the 1960s and they differ in temporal and spatial discretization, processes described, and in the constituent equations used. However, there are also similarities that have consequences for the way these models are commonly applied. All models of watershed hydrology aggregate the real hydrologic system on a particular element scale in space and time. The spatial scale might vary from grid cells of tens of meters to models that treat the whole catchment as a single unit, and the temporal scale might vary, for example, from 15-minute intervals to monthly time steps. It is common to assume homogeneity of processes or watershed characteristics on scales smaller than the one applied, that is, parameters are assumed to be effective on a certain scale (Fig. 1), although the heterogeneity of the real world on smaller scales is sometimes described by distribution functions (2). The characteristics of each model element, for example, storage or infiltration capacity, are described by parameters within the model. A common problem is that the scale on which these characteristics can be measured are usually different, mostly smaller, than the scale of the model element. The effect of this difference between model and measurement scale is that one has to revert to alternative methods to estimate model parameters. The usual approach is to observe the responses of the real hydrologic system, for example, streamflow, and compare them to predictions of the model. The modeler then adjusts the model parameters, in a process usually referred to as calibration, until model predictions and observations are as close as possible. Calibration can be performed using manual or automatic techniques and the available literature on this subject is immense. Between 3 to 10 years of observations are required for calibration, depending on the model complexity and the informational content of the data (3). Shorter periods can suffice when the data sufficiently trigger the response modes of the model (4).

Alternatives to model calibration have to be found when no or insufficient time series of the variable under investigation are available for this process. This is a common problem, even in countries that have extensive measuring networks such as the United Kingdom that has more than 1400 gauging stations (5). It is also possible that sufficiently long streamflow time series are available but that the modeling objective is the prediction of a different variable, for example, groundwater levels. Then, a calibration with respect to the variable under study

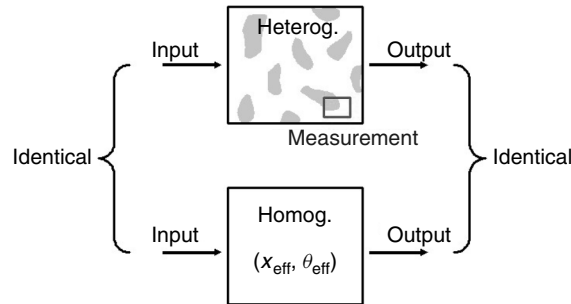


Figure 1. Definition of effective parameters and differences in scale between measurements and hydrologic modeling. (Modified from Reference 1).

would not be possible. With respect to the modeling of ungauged watersheds, there is also a difference whether the required predictions are limited to particular events or whether continuous simulation is required. Some reasonable estimates for event-based models can be derived, but this is not the case for continuous models where the predictions are highly uncertain (6). These two cases are therefore treated separately here.

MODELING THE EVENT-BASED WATERSHED RESPONSE

Empirical models are usually used for modeling individual events. In these models, the response characteristics of watersheds (e.g., mean annual flood or the percentage runoff) are related to watershed descriptors (e.g., area, drainage density, or dominant soil types) using regression equations (7;8, p. 301).

Another empirical approach that is very popular for event-based modeling in ungauged watersheds in many parts of the world is the curve number (CN) method. This approach was originally developed by the Soil Conservation Service for watersheds in the United States (9;10, pp. 147 et ff.). The basis of this technique is the assumption that the ratio of direct runoff to total precipitation is equal to the ratio of retained water to the potential maximum retention. The value of the potential maximum retention S can be calculated using values for CN:

$$S = \frac{1000}{\text{CN}} - 10 \quad (1)$$

where S is calculated in inches. Actual values for CN (0–100) given in tables or graphs are a function of soil type, land use, and antecedent moisture conditions (10, pp. 148–149), which makes this approach so attractive for modeling ungauged watersheds. These graphs and tables were originally derived from measured rainfall-runoff data on a small watershed or hillslope scale (8, p. 184).

These parametrically simple event-based models have been applied with some success (7,11). However, there is a trend to move from event-based models to those that provide continuous simulation because the initial conditions for event-based models are a major source of uncertainty. A recent workshop report on challenges in hydrologic predictability noted “in watershed rainfall-runoff transformation ... initial and boundary conditions are

the critical issues” (12, p. 17). This problem has led to a more holistic approach to flood management in some countries. In the United Kingdom, for example, there is a move to replace event-based modeling with continuous approaches (13,14).

MODELING THE CONTINUOUS WATERSHED RESPONSE

Several approaches to estimating parameters in ungauged watersheds are available for continuous simulation models. The two most common approaches are (1) the derivation of regression relationships between model parameters and watershed characteristics (5,6,15–17) and (2) estimation of parameter values from measurable watershed (mainly soil) properties (18–23).

In the first approach, a chosen model structure is calibrated to a large number of watersheds for which sufficiently long and informative observations are available. An attempt is then made to derive regression equations that predict its value using a combination of several watershed characteristics. A separate equation is commonly derived for each model parameter. The parameter values in the ungauged watershed can then be estimated using the derived equations and a prediction can be made. Figure 2 shows a typical procedure for extrapolating parameters from gauged watersheds using regression analysis (6). The steps are as follows (Fig. 2): (1) Select catchments and their characteristics. (2) Select and calibrate the local model structure. (3) Select and calibrate the regional model structure. (4) Predict flow at the ungauged site. Currently, this is probably the most often applied technique; however, Wagener et al. (6) found considerable uncertainty in typical predictions using this approach.

Sometimes, it might be possible to derive at least some of the model parameters directly from measurable watershed characteristics. The scale difference between model parameters and measurements might be relatively small for some parameters if the model uses a very fine spatial distribution, or simple equations can be used to derive these parameters from a combination of watershed characteristics. Koren et al. (22), for example, show how storage capacities can be estimated from soil properties such as field capacity and wilting point. These properties are usually derived from point samples analyzed on the laboratory scale. This makes using these values for lumped parameter estimation questionable because *there is generally no theory that allows the estimation of the effective values within different parts of a heterogeneous flow domain from a limited number of small scale or laboratory measurements* (8). On the other hand, this approach does not assume that all the model parameters are independent as in the earlier mentioned regression technique. The idea of Koren et al. (22) is therefore rather to derive good initial estimates for a subsequent calibration procedure in gauged watersheds, that is, to reduce the calibration effort, and also for ungauged watershed and distributed modeling approaches. Very few examples can be found in the literature where models, using only measured parameters, have been applied without further calibration. It is unlikely that reliable

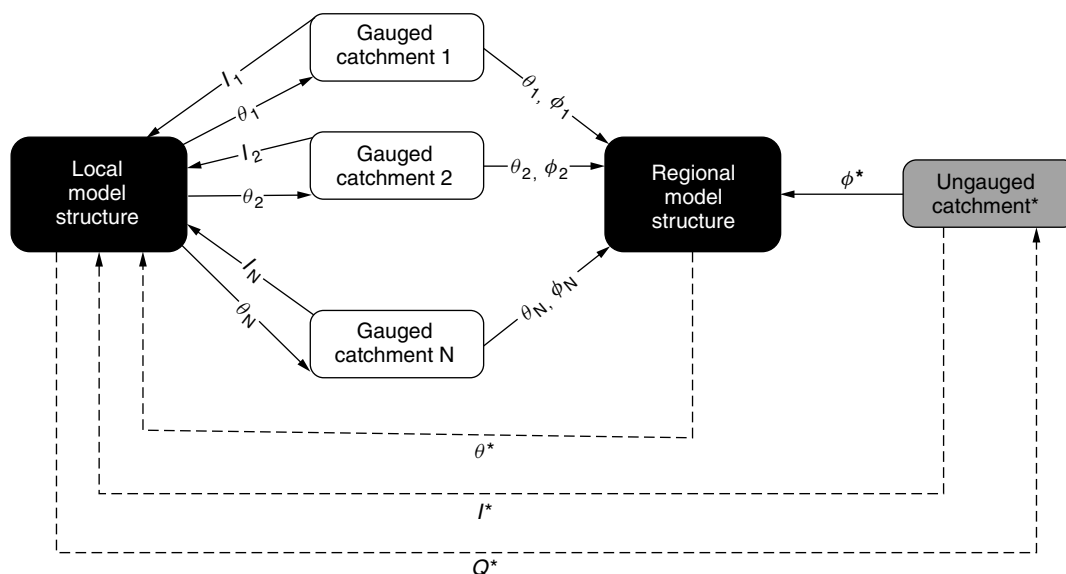


Figure 2. Schematic description of the statistical regionalization approach. (Modified from Reference 6).

predictions can be obtained by this approach from the current generation of model structures.

CONCLUSION

The ungauged problem is currently an area of extensive research and it can be expected that considerable progress will be made during the coming years (6,24,25). The complexity of the problem requires a holistic approach that can be provided only by a wide variety of hydrologists working on different topics. However, the potential value of the scientific outcome is very high. Current predictions in ungauged watersheds have to be considered as very uncertain though and must be used carefully in decision-making.

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CORPS TURNED NIAGARA FALLS OFF, ON AGAIN

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Niagara Falls, one of the Seven Natural Wonders of the World, attracts millions of tourists each year. But 30 years ago tourists flocked to see an unprecedented sight the mighty Falls reduced to a mere trickle by the engineering skill of Buffalo District.

Niagara Falls is 19 miles downstream from Lake Erie. Goat Island divides the Niagara River into two channels, separating Niagara Falls into American Falls on the American side and Horseshoe Falls on the Canadian side.

Public concern arose in the mid-1960s regarding the talus (rock fragments) accumulation at the base of the American Falls, which many felt detracted from its natural beauty. Three major rockfalls occurred in 1931, 1954 and 1959, depositing about 130,000 cubic yards of rock at the Falls' base.

In 1965, Congress authorized the U.S. Army Corps of Engineers to study the measures needed to preserve and enhance the beauty of the American Falls. The results of the Corps' study were submitted to the International Joint Commission (IJC) which, at the request of the American and Canadian governments, began an independent inquiry in 1967. The Commission established the American Falls Study Board, which included representatives from the Corps, Environment Canada, and eminent landscape architects to consider possible alternatives for preserving American Falls' future.

The IJC recommended to both countries that American Falls be temporarily dewatered to facilitate a timely and thorough study. The governments agreed, and on June 12, 1969, a rock cofferdam stopped the flow over American



The stark contrast between Niagara Falls with water and without actually increased tourist volume to the area

Falls. The Falls remained dry until November 25, 1969, when a backhoe removed the cofferdam.

Once stripped of its cascading white waters, the 1,100-foot-wide, 200-foot-high precipice revealed sights never before seen. One rock outcrop closely resembled a human profile. Another giant block of rock at the fall's base was larger than most houses in the city of Niagara Falls.

Special measures were taken to prevent damage to aquatic life in the American Falls channel. Terrestrial vegetation in the channel's small islands were protected and irrigated. Sprinklers kept Rochester shale wet on the face of the American Falls. In addition, railings at viewing areas were relocated and mass rock was stabilized to protect workers and the viewing public from injury.

While American Falls was dewatered, the board conducted a detailed geologic exploration. Forty-six core holes, totaling 4,882 feet, were bored. Dye and water testing on the completed holes identified weak points in the rock mass. Face mapping included topographic, stratigraphic and structural studies. Additional testing included terrestrial photogrammetry of the Falls' face, mapping rock fractures and joints, and measuring both



Workers were lowered by cranes, in safety cages, to inspect the face of the falls

water pressure in rock joints and horizontal movement in the adjacent rock mass.

"It was thrilling to walk on the American Falls riverbed shortly after dewatering and to view closely the huge exposed boulders," said Andrew Piacente, a civil engineering technician in the Water Control Branch. Piacente served as the district engineer's field representative to the American Falls Working Committee and the American Falls Board of Control during the project.

Talus studies involved examining the cobbles and boulders to determine their size, rock type and condition. The talus blocks were photographed and mapped. Talus depth ranged from 25 to 50 feet.

The board considered three alternative methods for removing all or part of the talus, and estimated the time and costs associated with each. The rock removal methods involved using either a cableway system between the U.S. and Canada, large cranes on and below the Falls' crest, or a large rock crusher together with a portable conveyor. The board determined a cableway with land disposal to be the most practical method.

Data gathered from the geologic exploration was used to build a realistic model of the American Falls 1/50th its actual size. The model, built by Ontario Hydro, included removable talus blocks, which allowed accurate simulation of various talus arrangements. The turbulence, mist, illumination, and volume of water were all closely duplicated.

Far from ruining the Falls' tourist appeal, the dewatering significantly increased tourist volume. People traveled thousands of miles to see the Corps unparalleled engineering feat.

"I've seen the Falls five or six times before, in winter and in summer," said a tourist quoted in the June 13, 1969 issue of the *Buffalo Courier Express*. "But this is the best time to see it. It's so unique."

Public opinion was considered very important. Public displays describing the Falls and the Board's undertakings were exhibited in both countries. The dewatering program received intensive national and international media coverage. Public opinion was strongly in favor of maintaining the Falls' natural appearance, leaving the accumulated talus fully intact.

In the end, the board agreed. After exhaustive deliberation, the IJC recommended that the U.S. and Canadian governments leaving the talus totally intact, considering the cost of removal, the irreversible nature of the project, and the resulting accelerated erosion. In addition, the IJC concluded that artificial means should not be employed to prevent further erosion of the Falls' crest.

The timeless beauty of majestic Niagara Falls continues to inspire and serve as a symbol of international cooperation between the U.S. and Canada. A 1975 IJC report to both governments stated that "consideration of the preservation and enhancement of the beauty of the American Falls cannot be limited to their physical aspects. The appeal and fascination of the Falls mean different things to different people. Their beauty is in the eye, the mind, and the heart of the beholder."

OPEN CHANNEL DESIGN

XING FANG

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An open channel flow is a flow having a free surface, which is subject to atmospheric pressure. This type of flow is typically found in rivers, creeks, irrigation canals, drainage conduits or ditches, culverts, spillways, and sanitary sewers. When water flows partially in a closed conduit (e.g., sewer pipe), this flow should be treated as an open channel flow. Open channel design is to design a channel to carry a certain amount of flow from one location to another under any provided channel geometry and topographic conditions. Design discharge that a channel needs to convey/carry is predetermined by a hydrological analysis. The analysis could be a simple application of the rational method ($Q = CIA$, where Q is the peak discharge, C is the watershed runoff coefficient, I is the average design rainfall intensity, and A is the upstream watershed area) or a complex watershed hydrological modeling, which may include rainfall design hyetograph, rainfall loss estimation, transformation of rainfall excess into runoff, and hydrograph routing at rivers detention/retention ponds, and reservoirs.

Parameters to define channel geometry include channel shape, cross-sectional area (A), wetted perimeter (P), hydraulic radius (R), top width (T), and hydraulic depth (D). Table 1 gives proprieties for common geometric channel shapes. Side slope of a channel (z, z_1, z_2 in Table 1)

Table 1. Geometric Properties of Different Channel Cross Sections

Channel Definition	Area (A)	Wetted Perimeter (P)	Hydraulic Radius (R)	Top Width (T)	Hydraulic Depth (D)
Rectangle	By	$b + 2y$	$\frac{by}{b + 2y}$	b	y
Trapezoid with equal side slopes	$(b + zy)y$	$b + 2y\sqrt{1 + z^2}$	$\frac{(b + zy)y}{b + 2y\sqrt{1 + z^2}}$	$b + 2zy$	$\frac{(b + zy)y}{b + 2zy}$
Triangle with equal side slopes	zy^2	$2y\sqrt{1 + z^2}$	$\frac{zy}{2\sqrt{1 + z^2}}$	$2zy$	$0.5y$
Trapezoid with unequal side slopes	$by + 0.5y^2(z_1 + z_2)$	$b + y(\sqrt{1 + z_1^2} + \sqrt{1 + z_2^2})$	$\frac{by + 0.5y^2(z_1 + z_2)}{b + y(\sqrt{1 + z_1^2} + \sqrt{1 + z_2^2})}$	$b + y(z_1 + z_2)$	$\frac{by + 0.5y^2(z_1 + z_2)}{b + y(z_1 + z_2)}$
Triangle with unequal side slopes	$0.5y^2(z_1 + z_2)$	$y(\sqrt{1 + z_1^2} + \sqrt{1 + z_2^2})$	$\frac{0.5y^2(z_1 + z_2)}{y(\sqrt{1 + z_1^2} + \sqrt{1 + z_2^2})}$	$2(z_1 + z_2)$	$0.5y$
Circle	$(\theta - \sin \theta)d_0^2/8$	$0.5\theta d_0$	$\left(1 - \frac{\sin \theta}{\theta}\right)d_0/4$	$d_0 \sin(0.5\theta)$ or $2\sqrt{y(d_0 - y)}$	$\left(\frac{\theta - \sin \theta}{\sin \theta/2}\right)d_0/8$
Parabola	$2/3 Ty$	$T + \frac{8y^2}{3T}$	$\frac{2T^2y}{3T^2 + 8y^2}$	$\frac{3A}{2y}$	$2/3y$
Round cornered rectangle $Y > r$	$\left(\frac{\pi}{2} - 2\right)r^2 + (b + 2r)y$	$(\pi - 2)r + b + 2y$	$\frac{\left(\frac{\pi}{2} - 2\right)r^2 + (b + 2r)y}{(\pi - 2)r + b + 2y}$	$b + 2r$	$\frac{(\theta/2 - 2)r^2}{b + 2r} + y$
Round-bottomed triangle	$\frac{T^2}{4z} - \frac{r^2}{z}(1 - z \cot^{-1} z)$	$\frac{T}{z}\sqrt{1 + z^2} - \frac{2r}{z}(1 - z \cot^{-1} z)$	$\frac{A}{P}$	$2[z(y - r) + r\sqrt{1 + z^2}]$	$\frac{A}{P}$

Symbol definition: y is depth; b is bottom width; z , z_1 , and z_2 are side slopes of horizontal versus vertical; d_0 is diameter; and r is radius for rounded corners of rectangle and triangle. Angle $\theta = 2 \cos^{-1}[(d_0 - 2y)/d_0]$, which ranges from 0 to 360 degrees.

is given as the horizontal increase with a unit vertical increase. Except for rectangular channels, hydraulic depth is not equal to the water depth y (Table 1) because the depth of water is variable in each cross section along flow lines. Hydraulic depth or mean depth (A/B) is therefore used and computed by dividing the cross-sectional area (A) by the width of the free surface (i.e., top width B). Wetted perimeter (P) is the total length of the channel boundary at a section wetted by the flowing liquid, and hydraulic radius (R) is determined by dividing channel cross-sectional area (A) by wetted perimeter (P). For very wide natural rivers (top width B is much larger than water depth D), hydraulic radius is approximately equal to mean depth D .

The flow in an open channel is uniform if the depth of flow does not vary along the length of the channel. The volumetric flow rate (discharge in ft^3/sec or m^3/sec) in a uniform open channel flow can be determined from the Continuity (conservation of mass) Equation 1 and Manning's Equation 2:

$$Q = VA \quad (1)$$

$$V = \frac{m}{n} R^{2/3} S_0^{1/2} \quad (2)$$

where V is the mean velocity of flow at a cross section, $m = 1.0$ ($\text{m}^{1/3}/\text{s}$) in SI units or $m = 1.49$ ($\text{ft}^{1/3}/\text{s}$) in English (British Gravitational) units, n is the Manning's roughness coefficient (dimensionless), R is the hydraulic radius, S_0 is the channel bed slope, and A is the cross-sectional area. Typical values of n for various channel conditions are given in Table 2. For channels with different roughness

Table 2. Typical Manning's n Coefficients (1)

Type of channel	Minimum	Normal	Maximum
Riveted and spiral steel	0.013	0.016	0.017
Coated cast iron	0.010	0.013	0.014
Uncoated cast iron	0.011	0.014	0.016
Galvanized wrought iron	0.013	0.016	0.017
Black wrought iron	0.012	0.014	0.015
Corrugated metal	0.021	0.024	0.030
Glass	0.009	0.010	0.013
Cement mortar	0.011	0.013	0.015
Finished concrete	0.010	0.012	0.014
Concrete culvert, straight	0.010	0.011	0.013
Concrete culvert with bends, connections	0.011	0.013	0.014
Concrete sewer with manholes, inlets, etc.	0.013	0.015	0.017
Wood stave	0.010	0.012	0.014
Clay drainage tile	0.011	0.013	0.017
Brick work	0.012	0.015	0.017
Earthen channel, straight, clean	0.017	0.020	0.025
Channel, straight with short grass, few weeds	0.022	0.027	0.033
Natural creeks and small streams (clean, straight, full stage, no rifts or deep pools)	0.025	0.030	0.033
Major streams (top width at flood stage greater than 100 ft)	0.025	—	0.060
Flood plains (pasture with high grass)	0.030	0.035	0.050

coefficients (n_i , $i = 1, \dots, N$) over the cross section (e.g., main channel and floodplain), a composite roughness (n_e) can be calculated by Equation 3 (2,3):

$$n_e = \left(\frac{\sum_{i=1}^N P_i n_i^{3/2}}{\sum_{i=1}^N P_i} \right)^{2/3} \quad (3)$$

For a composite channel (not simple geometry in Table 1), one should apply Manning's equation to compute discharges for each subchannel and summarize them to obtain the discharge in the whole channel.

For open channel design, channels should be classified as nonerodible channels and erodible channels. Most nonerodible channels are constructed and lined by using concrete, rip-rap, interlocking blocks, geotextiles, and vegetation, which can withstand erosion satisfactorily under all operational velocities. Unlined channels formed in natural materials (e.g., sand, gravel, sandy loam, firm soil, stiff clay) are generally erodible, especially under high velocity as encountered, e.g., during floods. To prevent the overtopping of channel lines by surface waves and surges, a *freeboard* is recommended; it is defined as the vertical distance from the top of the channel to the water surface at the design discharge. An average freeboard from 1.4 ft to 3.6 ft for discharges from 10 to 3000 ft³/sec, respectively, is recommended by the U.S. Bureau of Reclamation (4). Number of economical factors, for example, cost for land requisition controlled by channel top width, cost for lining channel controlled by wetted perimeter and materials, and cost of excavation controlled by channel area, can affect final selection of channel geometry.

In open channel design, the dimensions of a channel are computed for uniform flow; and Manning's equation (Equation 2) is applied after channel cross section (e.g., rectangle or trapezoid) and design discharge are specified. For lined channel design, one typically designs the channel with the best hydraulic section (Table 3), where the channel section has the least wetted perimeter for a given cross-sectional area and the channel is most hydraulic efficient. The best hydraulic section may not give the most economical channel because it could result as a very wide channel with a higher cost for land requisition. The semicircle has the least perimeter among all sections with the same area, but it may not be the most practical to construct with conventional materials. If one wants to

design a concrete trapezoidal channel capable of carrying 500 cfs with a channel slope of 0.0002 ft/ft. If one uses the best hydraulic section for trapezoidal channel given in Table 3 and applies Manning's equation Equation (2) as given below, water depth (y) is determined as 8.8 ft and channel width is 10.1 ft:

$$500 \text{ cfs} = \frac{1.49}{0.015} \left(\sqrt{3}y^2 \right) \left(\frac{y}{2} \right)^{2/3} (0.0002)^{1/2} \quad (4)$$

If a channel is placed in erodible material, the permissible velocity method can assure channel stability. For a trapezoidal unlined channel design, approximate permissible side slopes for various materials are given in Table 4 and maximum permissible velocities and roughness coefficients n are given in Table 5. One may use the best hydraulic section concept as the first estimate of the geometry for an erodible channel by using permissible side slope. If the velocity of the best hydraulic section at the design discharge is greater than the maximum permissible velocity, one needs to redo the design by using the maximum permissible velocity as the design velocity.

Table 4. Channel Side Slopes for Various Kinds of Materials (1)

Materials	Side Slopes (Horizontal:Vertical)
Rock	Nearly vertical
Muck and peat soils	1/4:1
Stiff clay or earth with concrete lining	1/2:1 to 1:1
Earth with stone lining	1:1
Firm Clay	1 1/2:1
Loose Sandy soil	2:1
Sandy Loam	3:1

Table 5. Maximum Permissible Velocities and Manning's n Values for Common Channel Materials (1,5)

Materials	V (ft/s)	Manning's n
Fine sand	1.50	0.020
Sandy loam	1.75	0.020
Silt loam	2.00	0.020
Firm loam	2.50	0.020
Stiff clay	3.75	0.025
Fine gravel	2.50	0.020
Coarse gravel	4.00	0.025

Table 3. Best Hydraulic Sections (1)

Cross Section	Area A	Wetted Perimeter P	Hydraulic Radius R	Top Width T	Mean Hydraulic Depth, D
Trapezoid (half of hexagon)	$\sqrt{3}y^2$	$2\sqrt{3}y$	$1/2 y$	$4/3 \sqrt{3}y$	$3/4 y$
Rectangle (half of a square)	$2y^2$	$4y$	$1/2 y$	$2y$	y
Triangle (half of a square)	y^2	$2\sqrt{2}y$	$1/4 \sqrt{2}y$	$2y$	$1/2 y$
Semicircle	$0.5\pi y^2$	πy	$1/2 y$	$2y$	$0.25\pi y$
Parabola $T = 2\sqrt{2}y$	$4/3 \sqrt{2}y^2$	$8/3 \sqrt{2}y$	$1/2 y$	$2\sqrt{2}y$	$2/3 y$

Note: y = Maximum water depth in cross section.

If one wants to design an earth trapezoidal channel using sandy loam and capable of carrying 500 cfs with channel slope of 0.0002 ft/ft, first design the channel for a best hydraulic cross section, which means that the hydraulic radius is half of the water depth (y). For a sandy loam channel, the recommended side slope $m = 3$ and application of Manning's equation results in a water depth of 8.2 ft and a flow velocity of 2.55 ft/s, which is great the maximum permissible velocity ($V_{\max}$) for sandy loam (Table 5). One has to use the maximum permissible velocity method to redesign the channel cross section:

$$V = 1.75 = \frac{1.49}{n} R_H^{\frac{2}{3}} S^{\frac{1}{2}} = \frac{1.49}{0.02} R_H^{\frac{2}{3}} (0.0002)^{\frac{1}{2}}$$

$$\therefore R_H = 2.14 \text{ ft}$$

$$\therefore A = by + 3y^2 = Q/V = 500/1.75 = 285.7 \quad (5)$$

$$\therefore P = b + 2y\sqrt{m^2 + 1} = b + 2y\sqrt{10} = A/R_H$$

$$= 285.7/2.14 = 133.5 \quad (6)$$

Using a trial-and-error method to solve Equations 5 and 6, one can get $y = 2.27$ ft and $b = 119.0$ ft. The final channel cross section with a 1-ft freeboard gives:

$$P_{\text{tot}} = 119.0 + 2(3.27)\sqrt{10} = 139.7 \text{ ft (42.58 m)}$$

$$T = 119.0 + 2 \times 3 \times 3.27 = 138.6 \text{ ft (42.25 m)}$$

$$A_{\text{tot}} = 119.0(3.27) + 3(3.27)^2 = 421.2 \text{ ft}^2 \text{ (39.0 m}^2\text{)}$$

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ORGANIC COMPOUNDS AND TRACE ELEMENTS IN FRESHWATER STREAMBED SEDIMENT AND FISH FROM THE PUGET SOUND BASIN

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As part of the National Water-Quality Assessment (NAWQA) Program, the USGS is investigating contaminants in streambed sediment and aquatic organisms and



Urban stream



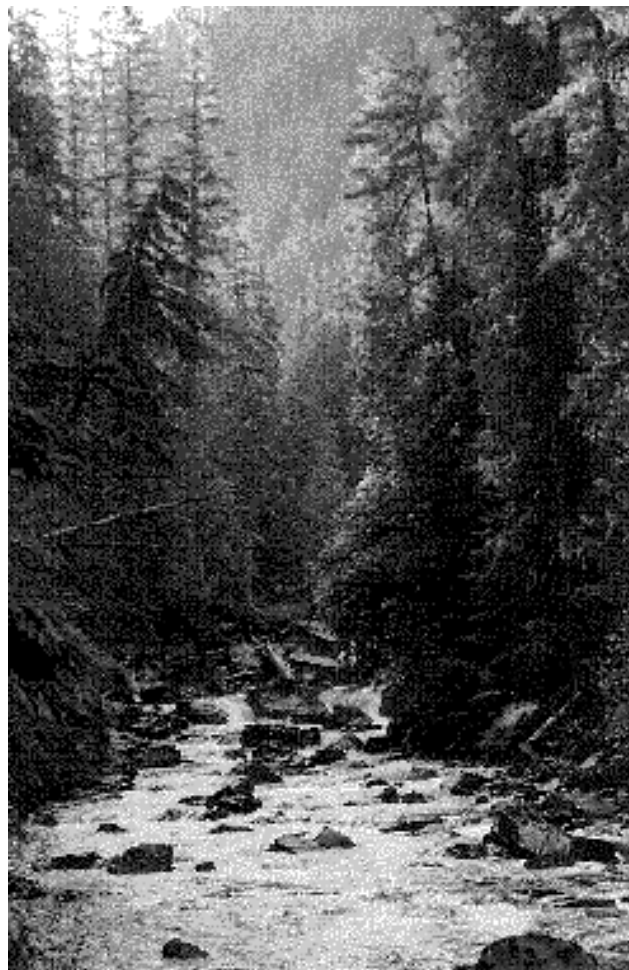
Agricultural stream

their relation to land use. One such study is being done in the Puget Sound Basin, which is located in northwestern Washington State and includes streams and rivers that drain to the Puget Sound, the Strait of Georgia, and the Strait of Juan de Fuca, but does not include marine waters. The basin encompasses 13,700 square miles; forest, urban, and agriculture are the principal land uses.

SUMMARY OF FINDINGS

Organochlorine Compounds—Highest concentrations of organochlorine compounds in streambed sediment were at an urban site on Thornton Creek near Seattle, where total chlordane, DDT, DDD, and DDE were found to exceed Canadian sediment quality guidelines. Concentrations are compared to Canadian guidelines because there are no sediment quality guidelines in the State of Washington.

Highest concentrations of organochlorine compounds in sculpin (a bottom-feeding fish) were found at the Thornton Creek site. Polychlorinated biphenyls (PCBs) and total DDT (DDT + DDD + DDE) exceeded New York State criteria for protection of fish-eating wildlife at Thornton Creek, and total PCB's exceeded these criteria at the West Branch Kelsey Creek at Bellevue, another urban site.



Forest stream

Concentrations are compared to New York State criteria because there are no criteria for protection of fish-eating wildlife in the State of Washington.

Effects—Elevated levels of organochlorine compounds such as DDT and PCBs are toxic to all animals and can bioconcentrate in tissue, cause tumors, and cause hormonal and behavioral problems. They can also suppress the immune and respiratory systems and cause abnormal development in aquatic species. The primary effect on aquatic communities is to reduce numbers of sensitive species, allowing species that are more resistant to contaminants to become dominant (Harte and others, 1991).

PAHs—Polycyclic aromatic hydrocarbon concentrations in streambed sediment exceeded Canadian guidelines in most urban streams and were highest at the West Branch Kelsey Creek at Bellevue.

Effects—Many PAHs, such as benzo(a)anthracene, benzo(a)pyrene, and chrysene, are carcinogenic, causing tumors in fish and other animals, and are acutely toxic to some organisms. Noncarcinogenic PAHs, such as fluoranthene, phenanthrene, and pyrene, are also toxic to some organisms. The effects on aquatic organisms of the PAHs found in sediment at Kelsey Creek are unknown, but concentrations of benzo(a)pyrene as high as those

observed in this study can cause precancerous tumors in fish (Eisler, 1987).

Trace elements—Concentrations of arsenic, cadmium, lead, mercury, and zinc frequently exceeded forest and reference conditions in streambed sediment and sculpin in urban streams.

Effects—Elevated levels of arsenic, cadmium, lead, mercury, and zinc may not be of concern in a naturally metal-rich region such as Puget Sound because the aquatic system has adapted to this type of environment, but excessive amounts of these elements can affect the nervous, respiratory, circulatory, and reproductive systems of aquatic organisms, as well as affect their development and feeding habits (Rand and Petrocelli, 1985).

DATA COLLECTION AND ANALYSIS

Streambed sediment and whole sculpin tissue were analyzed to assess the occurrence and distribution of contaminants and to better understand the fate of contaminants in the environment (Table 1). We collected samples in September 1995 from 18 sites, which were characterized on the basis of the predominant land use in the stream's basin—4 agricultural sites, 9 urban sites, 2 forest sites, and 3 reference sites, which are mostly forested and receive minimal impact from humans. At each site we collected the top 2–3 centimeters of streambed sediment in depositional areas; predatory bottom-feeding fish (sculpin) were collected from 17 of these sites. (See Crawford and Luoma, 1994, and Shelton and Capel, 1994, for a more complete description of the methods used.)

Fine-grained sediment and tissue accumulate trace elements and organic compounds associated with anthropogenic (human-related) activities. Sculpin are bottom-feeding fish that are not usually consumed by humans but are eaten by other fish and fish-eating wildlife. Organic compounds analyzed for were organochlorine pesticides, total PCBs, and other organic compounds (of the other organic compounds, only PAH values that exceeded Canadian guidelines are reported because they may have the most potential to harm aquatic and related organisms).

EVALUATION OF DATA

We compared our data to guidelines and criteria for organic compounds to show possible adverse effects to aquatic

Table 1. Contaminants Analyzed for in Streambed Sediment and Whole Sculpin Tissue from the Puget Sound Basin

Contaminant	Sediment ¹	Tissue
Organochlorine compounds	31 pesticides; total PCBs	26 pesticides; total PCBs
Other organic compounds	64	not analyzed ²
Trace elements	44	22

¹Finer than 2.0 millimeters for organic compounds, finer than 63.0 micrometers for trace elements.

²Tissue analysis too costly for this study.

organisms and fish-eating wildlife, and to local forest and reference conditions for selected trace elements to show possible effects of land use (Fig. 1).

Organic compounds—We compared levels of organochlorine compounds and PAHs detected in sediment to draft interim freshwater sediment quality guidelines developed by the Canadian Council of Ministers of the Environment (CCME). These guidelines were developed from toxicity and species abundance data for benthic organisms from studies throughout North America and represent total concentrations in sieved and unsieved sediment samples (CCME, 1995). The guidelines used are the threshold effects level (TEL), below which adverse effects to aquatic organisms are expected to occur **rarely**, and the probable effects level (PEL), above which adverse effects are predicted to occur **frequently**. Concentrations that exceed these guidelines may or may not have adverse effects on aquatic organisms; the comparisons should be used to indicate **potential** sediment quality problems that may warrant further study.

We compared concentrations of organochlorine compounds in sculpin to New York State Department of Environmental Conservation (NYSDEC) criteria (Newell and others, 1987). These criteria were determined from laboratory experiments using fish-eating wildlife and are considered one of the best sets of criteria for evaluating the effects of contaminated fish tissue on wildlife.

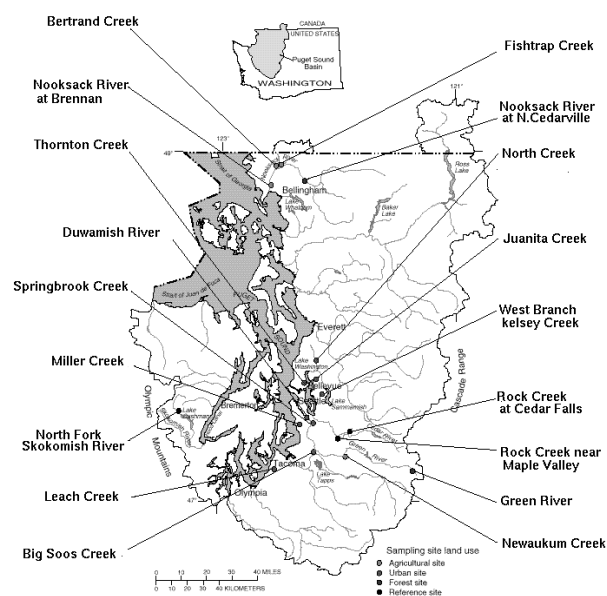


Figure 1. Puget sound basin.

Trace elements—We compared concentrations of selected trace elements in streambed sediment and sculpin

Figure 1 table. Organic compounds that exceed Canadian probable effects levels (PEL) and New York State Department of Environmental Conservation (NYSDEC) criteria, and trace elements that exceed median of forest and reference conditions at sampling sites in the Puget Sound Basin

Site	Sediment	Tissue
Bertrand Creek near Lynden	arsenic, cadmium, chromium, nickel, zinc	zinc
Nooksack River at Brennan	chromium, nickel, zinc	arsenic, chromium, lead, mercury, nickel, zinc
Thornton Creek near Seattle	arsenic, cadmium, chromium, lead, mercury, nickel, zinc, DDT	arsenic, lead, mercury, PCBs
Duwamish River at golf course at Tukwila	arsenic, cadmium, lead, zinc	arsenic, mercury
Springbrook Creek at Tukwila	arsenic, cadmium, chromium, lead, mercury, nickel, zinc	no tissue sampled
Miller Creek near Des Moines	arsenic, cadmium, chromium, lead, mercury, nickel, zinc	arsenic, lead, mercury
North Fork Skokomish River at Staircase Rapids	arsenic, chromium, nickel, zinc	arsenic
Leach Creek near Steilacoom	arsenic, cadmium, chromium, lead, nickel, zinc	arsenic, zinc
Big Soos Creek above hatchery near Auburn	arsenic, cadmium, chromium, lead, nickel	—
Fishtrap Creek at Flynn Road	arsenic, cadmium, chromium, lead, nickel, zinc	arsenic, mercury
Nooksack River at North Cedarville	nickel	—
North Creek below Penny Creek near Bothell	arsenic, cadmium, chromium, lead mercury, nickel, zinc	arsenic, mercury
Juanita Creek at La Juanita	arsenic, cadmium, chromium, lead, nickel, zinc	arsenic, cadmium, lead, mercury
West Branch Kelsey Creek at Bellevue	arsenic, cadmium, chromium, lead, mercury, nickel, zinc, PAHs	arsenic, cadmium, lead, mercury, zinc, PCBs
Rock Creek at Cedar Falls near Landsburg	arsenic, cadmium	chromium, nickel, zinc
Rock Creek near Maple Valley	cadmium, lead, zinc	mercury
Green River above Twim Camp	mercury	arsenic, nickel, zinc
Newaukum Creek near Black Diamond	cadmium, lead, zinc	—

tissue to median concentrations from the forest and reference sites. Land-use impacts may cause concentrations from the agricultural and urban sites to exceed these medians.

ORGANIC COMPOUNDS DETECTED IN STREAMBED SEDIMENT

Organochlorine pesticides were detected at 3 of the 18 sites sampled for streambed sediment: an agricultural site on Fishtrap Creek in the northern part of the basin, an urban site on Thornton Creek near Seattle, and a reference site on Rock Creek near Maple Valley. The highest concentrations were found at the urban site on Thornton Creek (Fig. 2).

PAHs were most frequently detected in streambed sediment samples from urban streams. The highest concentrations were found in the sample taken from West Branch Kelsey Creek at Bellevue (Table 2). (See also Tables 3–7.)

Definitions of organic compounds found in the Puget Sound Basin

DDT (dichlorodiphenyltrichloroethane) is an organochlorine insecticide banned from use in the U.S. in 1972. Total DDT refers to the sum of DDT and its breakdown products DDE and DDD.

Chlordane is an organochlorine insecticide banned from use in the 1980's. Total chlordane refers to the sum of *cis*-chlordane, *trans*-chlordane, *cis*-nonachlor and *trans*-nonachlor.

HCB (hexachlorobenzene) is a fungicide used as a seed and soil treatment, restricted from use in the 1980's.

Dieldrin is an organochlorine insecticide with restricted use in the U.S. since the 1970's.

Heptachlor epoxide is a breakdown product of the organochlorine insecticide heptachlor. It was used in the U.S. until the 1970's.

PCBs (polychlorinated biphenyls) are by-products of a variety of industrial products. Manufacture was stopped in the 1970's. There are over 209 breakdown products of PCBs, and total PCB refers to the sum of all forms detected.

PAHs (polycyclic aromatic hydrocarbons) are natural by-products of forest fires. Other sources include the steel and petroleum industry, the manufacture of coal tar and asphalt, power generation, burning trash, and vehicle emissions. Tons are emitted to the atmosphere and introduced to aquatic environments through oil spills and sewage discharge.

ORGANOCHLORINE COMPOUNDS DETECTED IN TISSUE

Total PCBs and/or at least 1 of 26 organochlorine pesticides were detected in tissue at 2 agricultural and 6 urban sites. The highest concentrations and greatest ranges of organochlorine compounds were detected at the Thornton Creek site (Fig. 3).

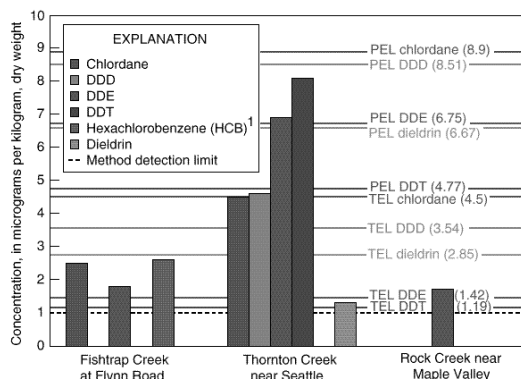


Figure 2. Concentrations of organochlorine pesticides in streambed sediment from selected sites in the Puget Sound Basin compared to Canadian criteria. (TEL, threshold effects level; PEL, probable effects level). ¹No published Canadian data for HCB.

Table 2. Concentrations of Polycyclic Aromatic Hydrocarbons (PAHs) in Streambed Sediment that Exceed Canadian Guidelines [values in micrograms per kilogram, dry weight; nd, not detected or below 50 micrograms per kilogram method detection limit; bold, above TEL; *italics*, above PEL]

Site Name	Benzo(a)anthracene ¹	Benzo(a)pyrene	Chrysene ¹	Fluoranthene ²	Phenanthrene ²	Pyrene ²
Fishtrap Creek	54	53	50	91	nd	87
Duwamish River	52	Nd	56	91	nd	79
Springbrook Creek	370	450	520	890	370	770
Juanita Creek	76	73	83	150	78	120
West Branch Kelsey Creek	<i>680</i>	<i>1700</i>	<i>950</i>	<i>2800</i>	<i>850</i>	<i>2300</i>
Leach Creek	57	62	65	100	51	94
Miller Creek	100	120	130	230	120	200
North Creek	Nd	Nd	nd	61	nd	56
Thornton Creek	220	310	270	470	200	410
Rock Creek near Maple Valley	270	Nd	200	320	150	240
Rock Creek at Cedar Falls	50	54	63	nd	160	nd

¹Weakly carcinogenic (Eisler, 1987).

²Noncarcinogenic.

Table 3. Organochlorine Compounds in Sediment from the Puget Sound Basin, September 1995 (micrograms per kilogram dry weight in less than 2 millimeter size fraction bottom material unless otherwise specified; MDL, method detection limit; nd, nondetect or below MDL; O, detection below MDL; E, detection level different from MDL, —, no published Canadian guidelines)

Site Type	MDL	Interim Canadian Sediment Quality Guidelines, Threshold Effects Level ¹	Interim Canadian Sediment Quality Guidelines, Guideline Effects Level ¹	Agricul-tural	Agricul-tural	Agricul-tural	Agricul-tural	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Forest	Reference	Reference	North Fork
USGS site number																				
Aldrin	1.0	—	—	12212500	12212100	12108500	12213140	12112600	12113390	12113375	12120490	12119850	12091300	12103326	12125900	12128000	12210700	12118500	12113375	1206495
Chloronel	5.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Dacthal (DCPA)	5.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Dieldrin	1.0	2.85	6.67	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	1.3	nd	nd	nd	nd
Endrin	2.0	2.67	62.4	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Heptachlor	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Heptachlor epoxide	1.0	0.6	2.74	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Hexachlorobenzene	1.0	—	—	nd	2.6	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Isodrin	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Lindane	1.0	0.94	1.38	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
(gamma-BHC, gamma-HCH)																				
Mirex	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Oxychlorodane	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Pentachloroisole	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
PCB	50	34.1	277	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Toxaphene	200	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Endosulfan	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
alpha-BCH	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
beta-HCH (beta-BHC)	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	1.6	nd	nd	nd	nd
cis-Chlordane	1.0	4.5	8.9	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
cis-Nonachlor	1.0	—	—	nd	2.5E	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
cis-Permethrin	5.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
o,p'-DDD	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
o,p'-DDE	1.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
o,p'-DDT	2.0	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd

(continued overleaf)

Table 3. (Continued)

Site Type	MDL	Interim Canadian Sediment Quality Guidelines, Probable Effects Level ¹	Interim Canadian Sediment Quality Guidelines, Probable Effects Level ¹	Agricul-tural	Agricul-tural	Agricul-tural	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Forest	Reference	Reference	Reference
<i>o,p'-Methoxychlor</i>	5.0	—	—														
<i>p,p'-DDD</i>	1.0	3.54	8.51														
<i>p,p'-DDE</i>	1.0	1.42	6.75														
<i>p,p'-DDT</i>	2.0	—	—														
<i>p,p'-Methoxychlor</i>	5.0	—	—														
<i>trans-Chlordane</i>	1.0	—	—														
<i>trans-Nonachlor</i>	1.0	—	—														
<i>trans-Permethrin</i>	5.0	—	—														
organic carbon (grams per kilogram)				8.1	20	8.5	3.2	19	3.9	20	4.5	13	3.1	4.8	11	9.1	250 ²
inorganic carbon (grams per kilogram)				nd	nd	Nd	nd	nd	nd	nd	nd	nd	0.2	nd	nd	0.9	nd
organic + inorganic carbon (grams per kilogram)				8.1	20	8.5	3.2	19	3.9	20	4.5	13	3.3	4.8	11	10	250
particle size (percent < 62 micron)				5	9	7	8	19	5	26	5	10	5	9	4	7	33
duplicate size fraction (percent <62 micron)					8											6	4

¹CCME/Canadian Council of Ministers of the Environment, 1995, Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life: Winnipeg, Manitoba, report CCME EPC-98E, Task Group on Water Quality Guidelines, 38 p.

²The value for organic carbon is very high for this sample. The value has been verified by the U.S. Geological Survey National Laboratory, but may not represent typical conditions in this stream.

Table 4. Metals in Streambed Sediment from Sites in the Puget Sound Basin, September 1995 (micrograms per gram unless otherwise specified, in less than 63 micron size fraction dry weight; MDL, method detection limit; nd, not detected or below MDL; (), detection below MDL)

	Agricul- tural	Agricul- tural	Agricul- tural	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Forest	Reference	Reference	Reference
Site Type	MDL	WA	WA	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Rock Creek	at Cedar Falls	North Fork River
USGS site number	12212500	12212100	12108500	12213140	12112600	12113390	12120490	12119850	12091300	12103326	12125900	12128000	12103380	12210700	12118500	12117695	12117695	12056495
Aluminum (%)	4.5	5.4	6.2	8.4	6	7.1	7.3	6.9	6.3	7.2	5.8	6.7	7.5	8.7	3.9	7.2	7.9	
Antimony	1	1	0.5	0.7	1	1	1	1	1	2	2	3	0.5	0.6	1	0.6	0.6	
Arsenic	15	21	6.2	nd	23	12	12	16	10	15	33	19	2.5	7	5.3	7.9	7.7	
Barium	400	600	370	630	410	430	520	530	420	500	490	510	390	580	230	370	670	
Beryllium	1	2	1	2	1	1	1	1	1	1	1	1	1	1	1	1	2	
Bismuth	10	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Cadmium	0.1	0.6	0.4	0.2	0.3	0.6	0.4	1.5	0.7	0.8	0.7	1.4	0.2	0.1	0.7	0.3	0.2	
Calcium (%)	0.005	1.6	1.8	2.8	2	2.2	2.2	2.1	2.5	2.3	2.1	2.5	1.9	3.4	2.2	1.9	1.1	
Cerium	4	30	35	44	34	36	39	32	35	36	27	35	54	40	15	36	88	
Chromium	1	78	50	140	99	57	110	110	93	110	120	120	26	80	54	65	110	
Cobalt	1	23	26	24	16	17	17	18	15	16	15	18	17	21	8	17	22	
Copper	1	26	38	50	40	44	29	82	29	44	28	58	31	40	42	44	48	
Europium	2	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Gallium	4	12	10	13	12	15	14	14	11	14	12	13	19	18	8	14	19	
Gold	8	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Holmium	4	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Iron (%)	0.005	21	8.6	4	4.1	4.9	3.8	3.9	3.2	3.6	4.1	4.2	4.4	4.4	1.7	3.7	4.4	
Lanthanum	2	14	15	21	15	17	17	15	16	17	12	19	23	19	10	18	23	
Lead	4	12	30	8	18	33	43	110	71	73	46	190	7	nd	23	13	13	
Lithium	2	10	20	30	20	20	20	30	10	20	20	20	30	20	20	20	50	
Magnesium (%)	0.005	0.75	0.98	2.4	1.2	1.1	1.2	1.1	0.9	1.1	1	1.3	0.78	1.9	0.57	0.87	1.4	
Manganese	4	1300	9200	800	1500	1700	1500	2300	3800	930	2200	2100	850	750	570	1000	940	
Molybdenum	2	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0.07	
Mercury	0.02	0.06	0.09	0.06	0.08	0.1	0.08	0.34	0.08	0.15	0.18	0.53	0.34	0.06	0.11	0.1	0.07	
Neodymium	4	15	15	20	16	18	18	15	15	16	12	17	26	20	9	17	40	

continued overleaf

Table 4. (Continued)

	Agricultural	Agricultural	Agricultural	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Forest	Reference	Reference	Reference
Site Type	MDL	WA	WA	Lynden, WA	Flynn Road at Lynden, WA	Fishtrap Creek	Newaukum Creek	Nooksack River	Big Soos Creek Above Hatchery Near Auburn, WA	Duwamish River At Golf Course at Tukwila, WA	Juanita Creek at La Juanita, WA	West Branch Kelsey Creek Near Des Moines, WA	Miller Creek Bothell, WA	Creek Below Penny Creek Seattle, WA	Nooksack River at North Cedarville, WA	Rock Creek at Cedar Falls near Valley, WA	Skokomish River Staircase Rapids Hoodsport, WA	
Nickel	2	58	70	30	120	60	31	63	57	37	57	52	65	19	61	23	36	54
Niobium	4	nd	4	8	11	9	9	9	9	8	9	9	10	14	10	4	10	15
Phosphorus (%)	0.005	0.21	0.26	0.21	0.13	0.18	0.19	0.12	0.16	0.09	0.09	0.28	0.18	0.12	0.12	0.16	0.13	0.14
Potassium (%)	0.05	0.5	0.67	0.61	1.3	0.7	0.89	0.81	0.82	0.71	0.82	0.62	0.79	1	1.2	0.41	0.59	1.6
Scandium	2	11	13	14	19	13	15	16	15	13	15	12	15	20	17	7	13	17
Selenium	0.1	0.9	1.4	1.2	0.7	1	0.4	0.4	0.6	0.3	0.4	1.6	0.6	0.9	0.7	4.6	1.1	1.4
Silver	0.1	0.3	0.3	0.3	0.1	0.3	0.3	0.2	0.5	0.2	0.3	0.4	2.2	0.3	0.1	0.4	0.3	0.2
Sodium (%)	0.005	1.1	1.3	1	2.1	1.5	1.6	2	1.6	2	2	1.5	1.9	1.3	2.2	0.85	1.3	1.6
Strontium	2	230	240	210	480	240	300	310	260	310	330	250	310	180	630	190	220	190
Sulfur (%)	0.05	0.14	0.17	0.16	0.2	0.13	0.1	0.07	0.13	0.05	0.05	0.17	0.14	0.07	0.32	0.28	0.08	0.1
Tantalum	40	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Thorium	4	4.1	nd(3.1)	7.3	3.9	nd(4.5)	8.8	4.2	4.8	5.2	3.5	nd	3.8	4.9	4.1	nd	4.6	11
Tin	10	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Titanium (%)	0.005	0.32	0.37	0.38	0.53	0.37	0.45	0.44	0.42	0.42	0.44	0.33	0.44	0.56	0.5	0.18	0.44	0.49
Uranium	0.05	1.47	1.99	1.35	2.4	1.85	1.68	1.85	1.91	1.78	1.79	1.79	1.94	1.86	1.81	54	1.61	3.14
Vanadium	2	65	120	100	160	110	110	120	120	110	120	120	120	100	140	140	110	130
Yttrium	2	18	17	20	20	18	19	18	16	17	17	15	17	35	19	14	15	26
Ytterbium	1	2	2	2	2	2	2	2	2	2	2	1	2	3	2	1	1	1
Zinc	4	130	250	150	99	93	140	140	470	180	190	180	330	94	85	95	88	110
Organic + inorganic carbon (grams per kilogram)	0.01	5.68	6.62	8.21	1.19	6.66	3.68	2.9	6.37	2.44	3.07	10.7	5.11	5.43	0.84	22.7 ⁱ	8.89	3.91
Inorganic carbon (grams per kilogram)	0.01	0.17	0.17	0.04	0.01	0.03	0.02	0.02	0.03	0.03	0.01	0.06	0.03	0.03	0.02	0.05	0.05	0.01
Organic carbon (grams per kilogram)	0.01	5.51	6.45	8.17	1.18	6.63	3.66	2.88	6.34	2.41	3.06	10.1	5.08	5.4	0.82	22.2	8.84	3.9

The value for organic carbon is very high for this sample. The value has been verified by the U.S. Geological Survey National Laboratory, but may not represent typical conditions in this stream.

Table 5. (Continued)

Site Type	MDL	Canadian Interim Sediment Quality Guidelines for Polycyclic Aromatic Hydrocarbons (Threshold Effect Levels) ¹	Canadian Interim Sediment Quality Guidelines for Polycyclic Aromatic Hydrocarbons (Threshold Effect Levels) ¹	Agri-cultural	Agri-cultural	Agri-cultural	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Urban	Forest	Reference	Reference	Reference
Compound																					
4-Bromophenyl-phenylether	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
4-Chloro M-Cresol	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
4-Chlorophenyl-phenylether	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Acenaphthene	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Acenaphthylene	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Acridine	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Anthracene	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
9,10-Anthraquinone	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Azobenzene	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Benzo(a)anthracene	50	31.7	385	nd	26E	35E	31E	52	76	41E	680	57	100	35E	220	nd	nd	nd	nd	nd	
Benzo(a)pyrene	50	31.9	782	nd	53	nd	27E	45E	73	38E	130	28E	35E	31E	39E	nd	nd	nd	nd	nd	
Benzo(b)fluoranthene	50	—	—	nd	66	nd	41E	65	92	1700	84	120	49E	260	nd	nd	nd	nd	nd	nd	
Benzo(c)quinoline	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Benzo(g,h,i)perylene	50	—	—	nd	34E	nd	nd	37E	58E	440	43E	62E	35E	170	nd	nd	nd	nd	nd	nd	
Benzo(k)fluoranthene	50	—	—	nd	41E	22E	nd	25E	74	790	57	110	34E	230	nd	nd	nd	nd	nd	nd	
bis(2-Ethylhexyl) Phthalate	50	—	—	53	180E	67	61E	55	110	2400E	270	220	93E	990E	55	74E	1100E	51E	84	84	
Butylbenzyl Phthalate	50	—	—	44E	44E	43E	39E	44E	46E	87	42E	90	45E	110	46E	39E	nd	35E	50	50	
C8-Alkylphenols	50	—	—	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Carbazole	50	—	—	nd	15E	nd	20E	29E	38E	180	25E	65	38E	27E	63	nd	nd	nd	nd	nd	
Chrysene	50	57.1	862	24E	50	15E	25E	56	83	950	65	130	52	31E	270	26E	11E	200	63	nd	
Di-n-butyl Phthalate	50	—	—	73E	41E	44E	45E	60	62	48E	58	45E	52	48E	58	72E	42E	300	38E	53	
Di-n-octyl Phthalate	50	—	—	nd	nd	nd	nd	nd	nd	140E	55E	nd	nd	nd	nd	nd	nd	nd	nd	nd	
Dibenz(a,h)anthracene	50	—	—	nd	nd	nd	nd	nd	nd	280	nd	36E	nd	nd	93	nd	nd	nd	nd	nd	
Dibenzothiophene	50	—	—	nd	17E	nd	20E	nd	20E	63	21E	25E	29E	38E	nd	nd	nd	nd	nd	nd	

Table 7. Metals in Whole Sculpin Tissue from Sites in the Puget Sound Basin, September 1995 (microgram per gram or ppm dry weight; MDL, method detection limit; nd, nondetect or below MDL; 0, detection different from MDL)

[illegible]

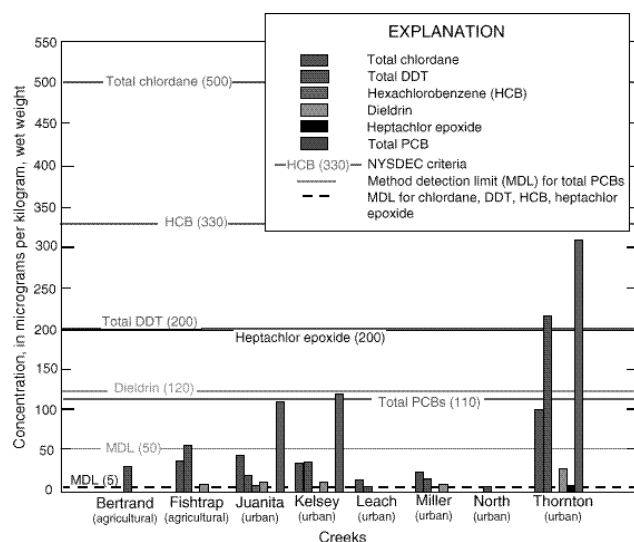
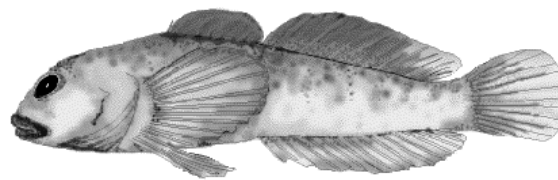


Figure 3. Organochlorine compounds in whole sculpin tissue from selected creeks in the Puget Sound Basin compared to New York State Department of Environmental Conservation (NYSDEC) criteria.



TRACE ELEMENTS DETECTED IN STREAMBED SEDIMENT AND WHOLE SCULPIN TISSUE

Concentrations of arsenic, cadmium, chromium, lead, mercury, nickel, and zinc were elevated in streambed sediment and sculpin from agricultural and urban sites compared to concentrations from the forest and reference sites. Arsenic and the heavy metals cadmium, lead, mercury, and zinc had the highest concentrations and the greatest range of concentrations at urban sites compared to those from the agricultural and the combined forest and reference sites, indicating possible enrichment of these elements in the urban areas (Fig. 4). The concentrations detected do not necessarily have negative impacts on the environment, but do suggest that land use may have led to increased levels of these elements.

READING LIST

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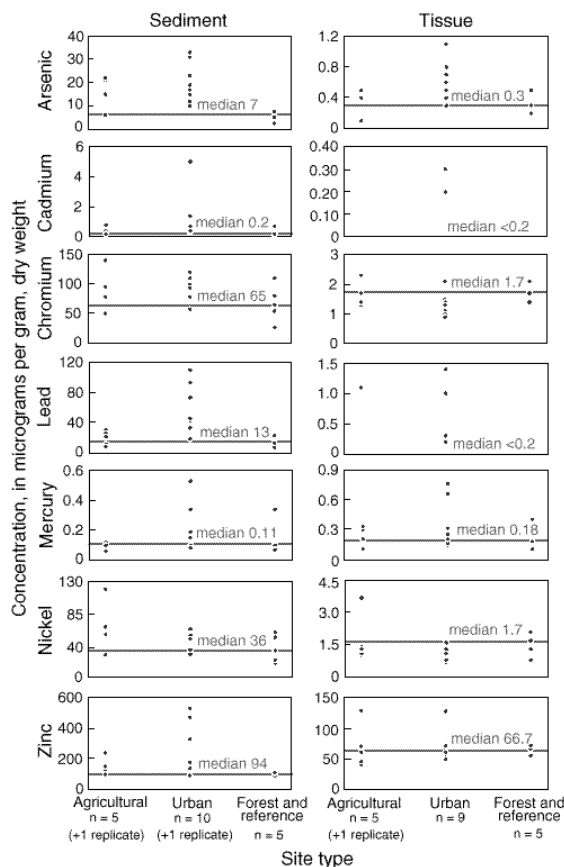


Figure 4. Concentrations of selected trace elements detected in streambed sediment and whole sculpin tissue collected in the Puget Sound Basin compared to the median of forest and reference site data. (n, number of samples; +1 replicate, one replicate sample data included; <, less than).

IMPERVIOUS COVER—PAVING PARADISE

SANDRA BIRD
U.S. Environmental
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THE IMPORTANCE OF IMPERVIOUS COVER

Nonpoint source pollution is pollution from diffuse sources such as urban/suburban areas and farmlands; this is now recognized as the primary threat to water quality in the United States. As urban and suburban development increases, the amount of land covered with impervious surfaces—areas where infiltration of water into the underlying soil is prevented—also increases. Roadways and rooftops account for the majority of this impervious area. Research in recent years has consistently shown a strong relationship between the percentage of impervious cover in a watershed and the health of the receiving stream. Scientists generally agree that stream degradation consistently occurs at even relatively low



Examples of impervious cover

levels of imperviousness (10 to 20%). Increased impervious surfaces alter stream hydrology resulting in lower flows during droughts and higher peak flows during floods. Roadways and other impervious areas channel pollutants directly into streams without their being processed during transport through the soil. With advance planning and identification of at-risk watersheds, total impervious cover can be reduced during development within a watershed, and steps can be taken to mitigate the impacts of added impervious cover.

GIS AND SPATIAL DATA

A Geographic Information System (GIS) is a computer system capable of assembling, storing, manipulating, and displaying geographically referenced information (i.e., data identified according to set locations). The way maps and other data have been stored or filed as layers of information in a GIS makes it possible to perform complex analyses.

With the aid of GIS software, a human analyst can efficiently and accurately identify and categorize ground features visible in computer-generated versions of aerial photographs that represent true map distances available throughout the nation. High-resolution satellite imagery is rapidly expanding use of remote sensing techniques for impervious cover estimation. Classified land cover data are now available throughout the country. Detailed road networks and block level census data are other GIS data sources available to aid in estimating impervious cover.

RESEARCH OBJECTIVE

Impervious cover is proposed as an indicator of aquatic conditions for sub-watersheds throughout the country. Researchers are focusing on methods that would be useful in doing region-wide environmental assessments. The usefulness of impervious cover as an indicator is a function of the ease and accuracy for estimating it. Testing is underway to determine with what degree of accuracy impervious cover can be estimated for sub-watershed areas from data available throughout a region.

RESEARCH SUMMARY

An impervious cover test data set for 56 sub-watersheds in Frederick County, MD was developed and used to evaluate different estimation techniques suitable for application to a regional-scale characterization. Using a combination of data sources, researchers were able to estimate the percentage of impervious cover in a watershed to within $\pm 1\%$.

Additional impervious cover data sets are being developed from aerial photographs for approximately 209 watersheds in the Atlanta, GA area for two time periods. These data sets will be used to confirm estimation methods and to determine the accuracy of methods to project future changes in impervious surface area. The estimation technique that was developed in this research project can be used by local and national agencies to target watersheds for monitoring and mitigation efforts.

PHYTOREMEDIATION BY CONSTRUCTED WETLANDS

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A constructed wetland is a wastewater treatment facility that duplicates the processes in natural wetlands. Constructed wetlands are complex, integrated systems in which water, plants, animals, microorganisms, and the environment interact to improve water quality. Phytoremediation, defined as cleansing of the environment using vegetation, is the primary water treatment process in constructed wetlands. The advantages of phytoremediation using constructed wetlands are that it is environmentally friendly, a sustainable technology, and cost-effective; the primary disadvantages are potentially incomplete removal of pollutants and pest attraction. Successful implementation of phytoremediation technology in constructed wetlands is a function of a number of inter-related biophysicochemical factors, including the types and fate of toxicants, characteristics of soil and sediment, plant species and plant uptake, hydrology and water flow, and microbial population. Hence, phytoremediation by constructed wetlands generally reflects an integrated multidisciplinary effort that combines plant biology, microbiology, surface chemistry, hydrology, and environmental engineering. The initial construction of wetlands, taking into account all these factors to design the most appropriate and effective method for removing target pollutants is labor-intensive and expensive, but low maintenance and operating costs make constructed wetlands a much more economically viable alternative to conventional physical and chemical treatments. Above all, the use of plants in environmental cleanup may guarantee a greener and cleaner planet for us and for future generations.

INTRODUCTION

The rising demands of modern day society for economic development and improved standards of living are leading to increased use of raw materials (1). The use of chemicals has played a key role in the development of agriculture, industry, housing, transport, textiles, and health. Their use has contributed significantly to the rise in the standard of living among populations around the world, but a major side effect has been the continuous release of both natural and man-made substances, including gases, heavy metals, volatile organic compounds, soluble organic compounds, suspended solids, and nitrogen and phosphorus compounds into the air, water, and terrestrial environments. Recent research is increasingly focused on the ecological and human health effects of contaminants, and guidelines or standards on soil, drinking water,

irrigation water, crop tissues, foodstuff, biosolids, and fertilizers are now becoming available (2). In addition to the human impact, some contamination problems result from geologic activity. For example, reports on reduced iron in Denmark, fluoride in Bavaria and Moldova, and arsenic in Bangladesh and India illustrate the fact that natural geochemical conditions may cause locally elevated concentrations of metals (1).

Environmental remediation technologies use physical, chemical, or biological processes that attempt to eliminate, reduce, isolate, or stabilize a contaminant or a multitude of contaminants (3). Depending on the technology used, the process may take place either at the location of the contamination (*in situ*), or the contaminated soil or water may be removed for *ex situ* treatment. The cost of environmental remediation using conventional techniques can be very high, a fact that has turned out to be the driving force in the development of new remediation technologies. Phytoremediation, defined as the cleansing of the environment using vegetation, has emerged as an effective, inexpensive, and environmentally friendly alternative for remediating contaminated soil and ground water (4). Commercial use of phytoremediation is currently rather limited; most of the technologies being developed are primarily at the experimental stage. The most developed and widely used phytoremediation technique is the use of constructed wetlands for treating wastewater from municipal sewage treatment facilities and industrial processing operations (5).

Wetlands prevent floods, store water for times of drought, and most importantly, clean silt and pollutants from runoff by trapping sediment and contaminants. By doing so, they prevent chemical and sedimentary pollution of rivers, lakes, and coastal waters; hence, environmentalists refer to wetlands as nature's kidneys (6). The physicochemical properties of wetlands provide many positive attributes for remediating contaminants (7). The rhizosphere of wetland vegetation provides an enriched culture zone for microbes involved in degradation (8). Furthermore, sediments or soils in wetlands display reducing conditions, which promotes certain remediation reactions (9). Chemical pollutants are also taken up by wetland plants, where they are accumulated and biotransformed to less toxic and/or immobile states (10) or are volatilized into the atmosphere (11). Plants also supply fixed carbon to sediments, which is an energy source for bacterial transformation of chemical contaminants (12). Constructed wetland ecosystems are typically characterized by carefully selected and induced biophysicochemical properties favorable for chemical, bio- and phytoremediation of contaminants, such as enhanced surface retention, complimentary redox conditions, hyperaccumulating vegetation, and degrading microbial population, and hence, provide conditions that help transform harmful pollutants to less harmful products or essential nutrients that can be used by the biota (13).

Although natural wetlands and terrestrial phytoremediation are traditionally used for removing wastes, phytoremediation using constructed wetlands is becoming more popular due to their efficiency in treating pollutants and cost-effectiveness. This article provides an overview

of the types of constructed wetlands, their advantages and disadvantages, the pollution removal mechanisms (with particular emphasis on phytoremediation attributes), and their effectiveness in removing organic pollutants, nutrients, pathogens, and heavy metals.

TYPES OF CONSTRUCTED WETLANDS

A constructed wetland is a designed, man-made complex of saturated substrates, emergent and submerged vegetation, animal life, and water that simulates natural wetlands for human use and benefits (14). Constructed wetlands offer an economical and largely self-maintaining alternative to conventional treatment of water contaminated by a variety of pollutants such as organic chemicals, toxic metals, and nutrients (14).

Wetland systems are typically described in terms of the position of the water surface (15). Accordingly, there are two basic types of constructed wetlands:

- subsurface flow systems
- free-water surface flow systems

Subsurface Flow Systems

A subsurface flow (SF) wetland is specifically designed for treating certain types of wastewater and is typically constructed as a bed or channel containing appropriate media (Fig. 1). Coarse rock, gravel, sand, and other soils have been used, but a gravel medium is most common in the United States and Europe (15). The medium is typically planted with the same types of vegetation present in marshes, and the water surface is designed

to remain below the top surface of the medium. The sizes of these systems range from small on-site units designed to treat septic tank effluents from individual homes, schools, apartment complexes, and parks that treat a few hundred gallons of wastewater per day to 1.5×10^7 liters per day systems that treat municipal wastewater (15). There are approximately 100 SF systems in the United States that treat municipal wastewater, the majority of these treat less than 3.8×10^3 m³/day (15). The most commonly used vegetation in SF wetlands includes cattail (*Typha* sp.), bulrush (*Scirpus* sp.), reed (*Phragmites* sp.), and sedge (*Carex* sp.). The submerged plant roots provide a substrate for microbial processes. Because most emergent macrophytes can transmit oxygen from leaves to roots, there are aerobic areas on the root surfaces. The remainder of the submerged environment in SF wetlands tends to remain devoid of oxygen (17). These systems are very useful in removing biochemical oxygen demand (BOD), total suspended solids (TSS), metals, and organic pollutants.

Free-Water Surface Flow Systems

Free-water systems (FWS) are designed to simulate natural wetlands, such as bogs, swamps, and marshes where water flows over the soil surface at a shallow depth (Fig.1). In FWS wetlands, water flows over a vegetated soil surface from an inlet point to an outlet point. The sizes of FWS wetlands systems range from small on-site units designed to treat septic tank effluents to large units that occupy more than 40,000 acres (15). Similar to SF systems, the most commonly used emergent vegetation includes cattail, bulrush, and reeds (17). The plant canopy formed by the emergent vegetation shades the water

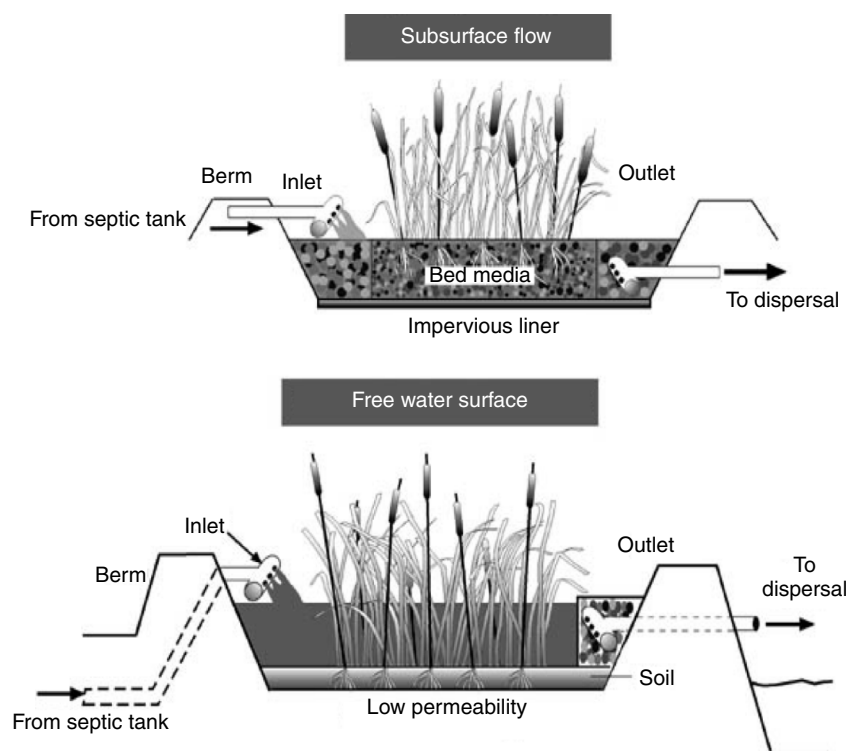


Figure 1. Subsurface flow and free-water systems (16).

surface, prevents the persistence and growth of algae, and reduces wind-induced turbulence in the water flowing through the system (15). Submerged portions of the living plants, the litter accumulated from the previous growth, and the standing dead plants provide a substrate for the microorganisms responsible for biological treatment in the system. Oxygen is available at the water surface and on living surfaces, allowing for aerobic activities. However, the bulk of the activities in the wetland is anaerobic (15). The low velocity and laminar flow of the influent in FWS systems result in very effective removal of TSS that contain BOD components, fixed forms of total nitrogen (TN) and total phosphorus (TP), trace levels of metals, and complex organics. The particulates are either oxidized or reduced and release soluble forms of BOD, TN, and TP, which are either adsorbed by soil/sediment or removed by microbial or plant populations in the wetland (15).

The use of FWS constructed wetlands has increased significantly since the late 1980s. SF wetlands are not now typically selected for larger flow municipal systems. The higher cost of the rock or gravel media makes a large SF wetland uneconomical compared to a FWS wetland despite the smaller area required for a SF wetland (15). Cost comparisons have shown that at flow rates above 60,000 gallons per day, it is usually cheaper to construct a FWS wetland system. However, there are exceptions where public access, mosquito, or wildlife issues justify selecting a SF wetland. Currently, FWS systems are widely distributed in 32 states in the United States (15).

ADVANTAGES AND DISADVANTAGES OF CONSTRUCTED WETLANDS

Constructed wetlands provide various advantages. This widens their appeal among professionals, from engineers and those involved in the workings of wastewater treatment facilities to the environmentalists and people concerned with recreation. Unlike some water issues in which the advantages to one group are disadvantages to another, the effective operation of constructed wetlands can provide broad benefits to a wide range of interests (18). Some of the more obvious benefits are listed here:

- Constructed wetlands are less expensive to build than other treatment options. There is no need for high cost, electromechanical equipment or for high consumption of electrical energy. If properly constructed and maintained, the wetlands provide efficient treatment of wastewaters (6).
- Operating and maintenance expenses (energy and supplies) are low. They require only periodic (rather than continuous) on-site labor (6).
- They are tolerant of fluctuating hydraulic and contaminant loading rates (19).
- They facilitate water recycling and reuse (15).
- Constructed wetlands have great tolerance for varying amounts of wastewater loading. The loading rates can be altered by varying livestock production levels, changing climatic conditions, and changing management (6).

- They can be aesthetically pleasing and may also provide additional benefits such as wildlife habitat, public recreation, environmental education, and groundwater recharge (6).

However, there are certain obvious disadvantages and negative side effects from the operation, maintenance, and effectiveness of constructed wetlands:

- Constructed wetlands require a continuous supply of water and also require water level management to sustain vegetation (6).
- They require more land area than the conventional treatment plants, if advanced treatment is desired (14).
- Surface wetlands attract mosquitoes and may cause other pest problems (14).
- Treatment efficiency may vary based on the effects of season and weather (6).
- High levels of ammonia can destroy vegetation in wetlands (6).
- The long-term effectiveness of constructed wetlands is not well documented. Wetland aging may contribute to a decrease in contaminant removal rates over time.

POLLUTANT REMOVAL MECHANISMS

Pollutant removal processes occur by interaction with wetland vegetation, the water column, and the wetland substrate. The processes may be physical, chemical, or biological (20).

Physical pollutant removal processes in wetland systems occur due to the presence of plant biomass and substrate media. Plants physically retard the pathways of sedimentation; the soil or gravel substrate acts as a filter bed, thereby aiding physical removal of suspended solids by straining (20).

Chemical reactions between inorganic pollutants can lead to their precipitation from the water column as insoluble compounds. Exposure to atmospheric gases and sunlight can lead to the breakdown of certain organic pollutants and destruction of pathogens. Antibiotic chemicals can also play a role in removing pathogens from wastewater (20).

Biological processes play a major role in removing pollutants in constructed wetland systems. Six major biological reactions have been identified as aiding the pollutant removal performance of wetlands, photosynthesis, respiration, fermentation, nitrification, denitrification, and biological phosphorus removal (7). A more detailed discussion of biological mechanisms, particularly in regard to bio- and phytoremediation, is presented in a later section.

As in any other biological wastewater treatment system, wetland process rates depend on various environmental factors, such as temperature, pH, oxygen availability, and hydraulic and pollutant loads (21,22). Under some environmental conditions, process rates may slow down or cease altogether, or even reverse, releasing pollutants. In general, the effectiveness of pollutant removal processes

that rely on biological activities may be reduced due to a decrease in metabolic activities caused by low temperature; many metabolic activities are less effective if the pH is either too high or too low (20). Hydraulic overloading occurs when the flow exceeds the design capacity, thus reducing the actual hydraulic retention time. Pollutant overload occurs when the influent pollutant loads exceed the process removal rates; for example, nutrient overloads can cause excess algal bloom leading to eutrophication, and decomposition of excessive organic matter can consume dissolved oxygen resulting in anaerobic conditions (20–22). Nutrient and organic matter overloading may even reverse many pollutant removal processes (20).

PHYTOREMEDIATION ATTRIBUTES OF CONSTRUCTED WETLANDS

Phytoremediation using wetlands is a potentially economical remediation alternative that plays an important role in sustaining and restoring the environment. It has been an important aspect of constructed wetlands, which have been used successfully to detoxify large volumes of wastewater that contain dilute concentrations of contaminants. Pollutants in wastewater are generally removed by plants in a symbiotic association with beneficial soil bacteria (23). For many years, scientists and engineers have observed improvement in the quality of water flowing through natural wetlands. This knowledge is of invaluable importance in designing the most effective constructed wetland systems. However, assessing the phytoremediation potential of constructed wetlands is still a rather complex issue due to variable conditions of hydrology, soil/sediment types, plant species diversity, and seasonal effects on plant growth and water chemistry (24). Williams (7) provides an analytical assessment of the phytoremediation attributes of constructed wetlands, as discussed following.

The physicochemical properties of wetlands provide many positive attributes for remediating contaminants (7). The expansive rhizosphere of wetland herbaceous shrub and tree species provides an enriched culture zone for microbes involved in degrading pollutants (8,25,26). Most wetland soil/sediment zones have strongly reducing conditions (9). Remediation reactions that require reducing conditions are enhanced under the low redox potential of wetland systems (27).

However, anaerobic reducing conditions also decrease the depth to which plant roots penetrate, restricting plant uptake and rhizosphere actions to shallower aquifer levels (7). Zones of wetlands interfacing with flowing water systems can provide conditions for aerobic oxidation-dependent degradation reactions. Redox conditions within rhizospheres can also be altered by oxygen delivery by plant roots (28).

The oxidation states of wetlands are also strongly related to hydrology. The rise and fall of the water table can greatly alter redox zones (7). Soil porosity, percolation rates, and horizontal hydrologic conductivity rates control the rate of oxygen delivery to subsurface layers. Slower vertical and horizontal flow rates with high microbial activity quickly produce reducing conditions.

Conversely, more rapid subsurface flows increase oxygen delivery, thereby slowing the decline in redox (7). Plant roots can transport oxygen to the surrounding rhizosphere (28–30) along with plant tissue air spaces connected to belowground biomass (31). Complex interactions among the microbial community, plant species composition, soil/sediment characteristics, and hydrology create a complex set of remediation reactions in wetland ecosystems (7).

Phytoremediation reactions are influenced by seasonal variability of physical and chemical characteristics in response to terrestrial and climatic events (7,32). Plant photosynthesis and standing crop biomass exhibit dramatic seasonal changes in nearly all tidal freshwater vegetation communities (9). Long-term ecological succession can shift plant species composition in the wetlands thereby altering phytoremediation characteristics because phytoremediation by one wetland community may not be the same as that which the new successional plant community develops (7). Hence, successful long-term phytoremediation must carefully consider the management implications of secondary successions to avoid ecological shifts away from optimal plant community structure (33).

Given all the factors that can influence the effectiveness of a wetland system and keeping in mind that specific requirements vary depending on the functional role of wetland plants in the treatment systems, broad generalizations can still be made about the characteristics of plant species that are most suitable for constructed wetland wastewater treatment systems (34):

- ecological acceptability, that is, no significant weed or disease risks or danger to the ecological or genetic integrity of surrounding natural ecosystems;
- tolerance to local climatic conditions, pests, and diseases;
- tolerance of pollutants and hypertrophic waterlogged conditions;
- ready propagation and rapid establishment, spread, and growth;
- high pollutant removal capacity, either through direct assimilation or storage, or indirectly by enhancing microbial transformations.

Cattails (*Typha* sp.), bulrushes (*Scirpus* sp.), reeds (*Phragmites* sp.), and sedges (*Carex* sp.) generally meet these requirements and hence, are the typical macrophytes (rooted plants that anchor to the substrate) of choice in the majority of constructed wetland systems, including both subsurface flow and freshwater surface flow types (20).

EFFECTIVENESS OF PHYTOREMEDIATION IN WETLANDS

Constructed wetlands can remove a range of pollutants, including biodegradable organic substances, metals, and nutrients. Natural wetlands have lower efficiency in removing pollutants than constructed wetlands (10). The growth and adaptation of plants to anoxic conditions in wetland sediments drives many of the biological and chemical processes to transform the pollutants to a

variety of forms that differ in mobility and toxicity (12). For example, the activity of plant roots alters the surrounding conditions in the sediment, enhancing the rate of transformation and fixation of metals (35).

The following section, derived primarily from Williams (7) and Sundaravadivel and Vigneswaran (20), presents a review of the performance of both constructed and natural wetlands in regard to phytoremediation of the aforementioned three major classes of chemical pollutants.

Removal of Organics

Organic contaminants, such as trichloroethylene (TCE) and pesticides, have been remediated effectively by a variety of phytoremediation actions of terrestrial and wetlands species (7). Cunningham et al. (36) identified the following phytoremediation actions for successfully removing organic contaminants: accumulation into biomass, phytovolatilization, cellular degradation, and rhizosphere degradation. Hybrid poplars have had increasing success in phytoremediating TCE (37) and other volatile organic compounds (VOCs). Several VOCs, including TCE, were removed from surficial groundwater and subsequently degraded by poplar (*Populus deltoids* x *trichlocarpa*) test plots near toxic disposal sites in Maryland (38). Nietch et al. (39) experimented with bald cypress (*Taxodium distichum*) to remediate TCE and found that the TCE flux through bald cypress into surrounding air space decreased from day to night and from August to December. Much of this flux was controlled by evapotranspirational seasonality.

Nzungung et al. (40) summarized phytoremediation research on halogenated organic chemicals (HOC) by *Elodea canadensis*, *Myriophyllum aquaticum*, *Salix nigra*, and *Populus deltoides* and identified four major phytoremediation processes: (1) rapid sequestration by partitioning to lipophilic plant cuticles; (2) phytoreduction to less halogenated metabolites; (3) phytooxidation to haloethanols, haloacetic acids, and unidentified metabolites; and (4) assimilation into plant tissues as nonphytotoxic products. Additional types of VOC, semivolatile, and nonvolatile compounds were evaluated for their phytoremediation potential by hybrid poplar (41). Burken and Schnoor (41), developed the transpiration stream concentration factor (TSCF) and the root concentration factor (RCF) for 12 such compounds, including aniline, phenol, nitrobenzene, benzene, atrazine, and toluene. Newman et al. (42) reported phytoremediation via reductive dechlorination by *Populus trichocarpa* and emphasized the ecological concern of selecting a phytoremediating species that grows well in the local habitat, while not ecologically destabilizing existing communities. In another study, Ensley (43) reported that duckweed (*Lemna gibba*) reductively dechlorinates chlorinated phenols.

Removal of Metals

Wetland sediments are generally considered a sink for metals and, in the anoxic zone, may contain very high concentrations of metals in a reduced state (44). As such, the bioavailability of metals in wetlands is low compared to that in terrestrial systems with oxidized soils. Water-soluble and exchangeable metals are most bioavailable,

metals precipitated as inorganic compounds or metals complexed by humic substances or metals adsorbed onto hydrous oxides are potentially bioavailable, and metals precipitated as insoluble sulfides and metals bound to the crystal lattice of minerals are essentially unavailable (45). Because of reducing conditions, the depth of wetlands to which plants can penetrate is limited, and this restricts the uptake of contaminants and rhizosphere actions to shallower levels (7).

Understanding the role of phytoremediation in metal removal from wetlands has progressed over the years from basic uptake studies to quantification of chemical speciation and bioavailability and the role of genetic engineering in metal hyperaccumulation (46–50). In New Zealand, submerged, rooted aquatic species, such as *Ceratophyllum demersum*, were hyperaccumulators of arsenic (CF or concentration factor of 20,000) compared to cattail, *Typha orientalis* (CF of 100), a popular wetland plant species (46). Water hyacinths (*Eichhornia crassipes*), well known for their use in wastewater treatment, have also been found effective in accumulating heavy metals and in the uptake of pesticide residues, PCB, DDE, DDD, and DDT (47,51). Apparently, water hyacinths carried out phytoremediation by root sorption, concentration, and/or metabolic degradation (47). Duckweed (*Lemna minor*) and water velvet (*Azolla pinnata*) both effectively removed iron and copper at low concentrations; however, water velvet does not grow from May to October, and such production seasonality translates to a limited phytoremediation interval (52).

Removal of heavy metals by wetland hydrophytes presents the problem of plant biomass collection to prevent decomposition, thus recycling accumulated metals, because many of the wetland plant species typically undergo much faster decomposition compared to terrestrial species (7). Phytomining, the recovery of accumulated trace metals, might be side benefits of phytoremediation; however, concerns about plant matter getting into the food chain by direct consumption or by decomposition pathways (49) include considerable ecological and human health problems. Careful harvesting techniques must be used to prevent loss of metal-enriched plant biomass. Biomass recycling is a major concern because even wave action can remove fresh biomass prior to harvesting (7).

Removal of Nutrients

Removal of macronutrients, nitrogen and phosphorus, by stabilized constructed wetlands is a complex cyclic process believed to involve a number of "conceptual compartments," such as soils, sediments, and litter/peat (80% of the total nutrient load), water column (15–20%), and plants/other biota (5%) (20,53). The primary mechanisms of nutrient removal from wastewater in constructed wetlands are microbial processes such as nitrification and denitrification as well as physicochemical processes such as the fixation of phosphate by iron and aluminum in the soil filter (17). The hydraulic retention time, including the length of time the water is in contact with the plant roots, affects the extent to which the plant plays a significant role in the removal or breakdown of pollutants (17).

A survey of literature on the nutrient removal efficiencies of constructed wetlands indicates extreme variations (20). Phosphorus removal efficiency is strongly dependent on loading rate, 65–95% removal is achieved at loading rates less than 5 g/m²/y. However, removal efficiency decreases to 30–40% or even less when phosphorus loadings are greater than 10–15 g/m²/y (54). Nitrogen removal efficiencies reportedly decline at mass loading rates above 20 kg/ha/d (20). Total nitrogen removals up to 79% are reported at higher loading rates of up to 44 kg/ha/d from constructed wetland systems in the United States (55).

Nutrient removal by wetland systems is also a function of climate; tropical climatic conditions generally tend to increase nutrient removal efficiencies (20). The nitrogen and phosphorus removal efficiencies of a full-scale plant treating domestic wastewater in Bhubaneswar, India, are over 70% and 43%, respectively (56). A pilot scale study in Central India also indicated higher removal efficiencies for both nitrogen and phosphorus in the range of 58 to 65% (57). A full-scale, gravel bed, horizontal flow system build in Kathmandu, Nepal, reports nitrogen removal in the range of 80–99% and phosphorus removal from as little as 5% to 69% (58).

SUMMARY AND FUTURE DIRECTIONS

Constructed wetlands can be designed to remove a wide range of pollutants from wastewater. Due to their low operating, maintenance, and energy costs, constructed wetlands could rapidly become the ideal method to achieve sustainable wastewater management goals. Several laboratory-scale mesocosmic studies have established the versatile nature of wetlands to remove pollutants, but only a few large-scale field studies have been conducted to gauge the remediation potential of constructed wetlands. The use of constructed wetlands to treat wastewater is a relatively new and emerging technology. The initial results are encouraging, but the full range of capabilities and limitations of this technology have not yet been uncovered.

Degradation of pollutants by wetlands involves complex interactions among wetland species, sediment/water geochemistry, and climate. Therefore, it is crucial to determine the effectiveness of remediation by implementing a detailed monitoring program that accounts for the interactions of numerous variables ranging from the composition of the plant species to the seasonality of hydrologic flow. Carefully planned monitoring programs are essential to determine the effectiveness of the wetland remediation strategy and are also instrumental in achieving regulatory acceptance.

Public concern during the past 20 years has strengthened regulations on wastewater discharges and resulted in substantial progress in treating point sources of water pollution, especially for large cities and major industries. Widespread implementation of the constructed wetlands treatment technology may accomplish the same for small communities, small industries, farmers, and developers, where low cost technologies are important. The available knowledge on constructed wetlands has grown extensively

over the last decade, but much more research is needed to determine the capabilities of plants and microorganisms to remove pollutants from wastewater. As more wetland projects are undertaken and further field-based research is conducted, the treatment possibilities of constructed wetlands will be better understood.

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UNRECOGNIZED POLLUTANTS

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INTRODUCTION

The U.S. water pollution control program focuses on the control of a limited group of chemicals, the 126 current “priority pollutants.” The chemicals included on the list of priority pollutants were originally selected in the mid-1970s through a litigation settlement. The list did not receive appropriate peer review for its representation of the chemical contaminants of most pressing importance to public health and environmental

quality. That notwithstanding, the priority pollutants remain the focal point of water quality investigation and management. Limited attention is given to the evaluation and regulation of the many thousands of other potential pollutants that are in municipal and industrial liquid and solid wastes, even in light of the myriad of other chemicals currently in commercial and personal household use, and those that come into use annually.

Daughton (1) highlighted the growing concern about unrecognized, unregulated pollutants, indicating that over 22 million organic and inorganic substances exist, with nearly 6 million commercially available. The current water quality regulatory approach addresses fewer than 200 of these chemicals. He also noted,

“Since the 1970s, the impact of chemical pollution has focused almost exclusively on conventional “priority pollutants,” especially on those collectively referred to as “persistent, bioaccumulative, toxic” (PBT) pollutants, “persistent organic pollutants” (POPs), or “bioaccumulative chemicals of concern” (BCCs).

The “dirty dozen” is a ubiquitous, notorious subset of these, comprising highly halogenated organics (e.g., DDT, PCBs).

The conventional priority pollutants, however, are only one piece of the larger risk puzzle.

Regulated pollutants compose but a very small piece of the universe of chemical stressors to which organisms can be exposed on a continual basis.

Similarly, Dr. K. Hooper of the Hazardous Materials Laboratory, California EPA Department of Toxic Substances Control, recently noted (2),

“Over the past 25 years, tens of thousands of new chemicals (7 chemicals per day) are introduced into commerce after evaluation by USEPA. Few (100–200) of the 85,000 chemicals presently in commerce are regulated. We have reasons to believe that a much larger number than 200 adversely affect human health and the environment.”

Periodically, unregulated chemicals that are in use are discovered to be widespread pollutants, posing a significant threat to public health and the environment. Examples of such findings and illustration of the need to address the broader issue of previously unrecognized pollutants are presented below.

EXAMPLES OF PREVIOUSLY UNRECOGNIZED POLLUTANTS

Pharmaceuticals and Personal Care Products (PPCPs)

Increasing attention is being given to pharmaceuticals and personal care products (PPCPs) as environmental pollutants. At the California Bay Delta Authority (CBDA) Contaminant Stressors Workshop, Dr. Christian Daughton, Chief, Environmental Chemistry Branch, US EPA National Exposure Research Laboratory, addressed this issue in his presentation entitled, *“Ubiquitous Pollution from Health and Cosmetic Care: Significance, Concern, Solutions, Stewardship—Pollution from Personal Actions”* (1).

Daughton (1) pointed out that a wide variety of chemicals that are introduced into domestic wastewaters

are being found in the environment. Various chemicals (pharmaceuticals) used by individuals and to treat pets, outdated medications disposed of into sewage systems, and treated and untreated hospital wastes discharged to domestic sewage systems end up in receiving waters for domestic wastewater treatment plant effluents. Further, transfer of sewage solids ("biosolids") to land, industrial waste streams, landfill leachate, releases from aquaculture of medicated feeds, etc. also introduce these chemicals into the environment. Many of these chemicals are not new and have been present in wastewaters for some time. However, they are only now beginning to be recognized as potentially significant water pollutants and are largely unregulated as water pollutants.

According to Daughton (1),

"PPCPs [Pharmaceuticals and Personal Care Products] are a diverse group of chemicals comprising all human and veterinary drugs (available by prescription or over-the-counter; including the new genre of "biologics"), diagnostic agents (e.g., X-ray contrast media), "nutraceuticals" (bioactive food supplements such as huperzine A), and other consumer chemicals, such as fragrances (e.g., musks) and sun-screen agents (e.g., mehylbenzylidene camphor); also included are "excipients" (so-called "inert" ingredients used in PPCP manufacturing and formulation)."

Although the full range of impacts of PPCPs is just beginning to be investigated, PPCPs are being found to have adverse impacts on aquatic ecosystems. For example, Daughton (1) discussed the relationship between PPCPs and endocrine disrupters, which are believed to be responsible for causing sex changes in fish. In addition, in a feature article in *Environmental Science and Technology*, Eggen et al. (3) reviewed a number of the issues pertinent to understanding the impacts of PPCPs and other chemicals that can cause endocrine disruption, DNA damage/mutagenesis, deficiencies in immune system, and neurological effects in fish and other aquatic life. (Additional information on PPCPs is available at www.epa.gov/nerlesd1/chemistry/pharma/index.htm.)

Perchlorate

Perchlorate (ClO_4^-), used in highway safety flares and other applications, is another example of a chemical that is now being found in surface and groundwaters in sufficient concentrations to pose a threat to human health. Perchlorate is derived from several sources. Silva (4) of the Santa Clara Valley, CA Water District, noted the potential for highway safety flares to be a significant source of perchlorate contamination to water, even when the flares are 100% burned. According to him, one fully burned flare can leach up to almost 2,000 μg of perchlorate, and

"A single unburned 20-minute flare can potentially contaminate up to 2.2 acre-feet [726,000 gallons] of drinking water to just above the California Department of Health Services' current Action Level of 4 $\mu\text{g}/\text{L}$ [for perchlorate]" (4).

Silva also pointed out that more than 40 metric tons of flares were used/burned in 2002 alone in Santa Clara County. California's Office of Environmental Health

Hazard Assessment recently conducted an evaluation of the hazards of perchlorate in drinking water. The 4 $\mu\text{g}/\text{L}$ action level for perchlorate in drinking water was based on the detection limit; it has been revised to 6 $\mu\text{g}/\text{L}$ based on the recent OEHHA evaluation.

Polybrominated Diphenyl Ethers (PBDE)

Another unrecognized, unregulated pollutant is the polybrominated diphenyl ethers (PBDEs) used as a flame retardant on furniture and other materials. Hooper (2) recently discussed finding PBDE in human breast milk and in San Francisco Bay seals, and the fact that archived human breast milk shows that this contamination has been occurring for over 20 years. According to McDonald (5) of the California Environmental Protection Agency, Office of Environmental Health Hazard Assessment,

"Approximately 75 million pounds of PBDEs are used each year in the U.S. as flame retardant additives for plastics in computers, televisions, appliances, building materials and vehicle parts; and foams for furniture. PBDEs migrate out of these products and into the environment, where they bioaccumulate. PBDEs are now ubiquitous in the environment and have been measured in indoor and outdoor air, house dust, food, streams and lakes, terrestrial and aquatic biota, and human tissues. Concentrations of PBDE measured in fish, marine mammals and people from the San Francisco Bay region are among the highest in the world, and these levels appear to be increasing with each passing year."

PBDEs are similar to PCBs and are considered carcinogens. Some of the PBDEs are being banned in the United States and in other countries.

Pesticides

Another example of unidentified pollutants was given by Kuivila (6). She discussed the fact that approximately 150 pesticides used in California's Central Valley exist that are a threat to cause water quality problems in the Delta and its tributaries. The current pesticide water quality regulation program considers only about half a dozen of those.

CONCLUSIONS AND RECOMMENDATIONS

The presence of untold, unregulated pollutants in environmental systems, as illustrated above with the examples of PPCPs and others, is not unexpected based on the approach that is normally used to define constituents of concern in water pollution control programs. Based on the vast array of chemicals that are used in commerce, many of which are or could be introduced into aquatic systems from wastewater and stormwater runoff, it is likely that many other chemicals will be discovered in the future that are a threat to public health or aquatic ecosystems. A pressing need exists to significantly expand water quality monitoring programs to specifically search for new, previously unrecognized water pollutants. As demonstrated by the perchlorate and PBDE situations, monitoring programs that focus on priority pollutants stand to be significantly deficient in properly defining

constituents of concern with respect to impairing the beneficial uses of waters.

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POLLUTION OF SURFACE WATERS

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INTRODUCTION

According to the American College Dictionary, pollution is defined as “to make foul or unclean; dirty.” Water pollution occurs when a body of water is adversely affected by the addition of large amounts of materials to the water. When it is unfit for its intended use, water is considered polluted.

Comprising over 70% of the earth’s surface, water is undoubtedly the most precious natural resource that exists on our planet. Without the seemingly invaluable compound of hydrogen and oxygen, life on the earth would be nonexistent. It is essential for everything on our planet to grow and prosper. Although we as humans recognize

this fact, we disregard it by polluting our rivers, lakes, and oceans. We are slowly but surely harming our planet to the point where organisms are dying at a very alarming rate. In addition to innocent organisms dying off, our drinking water has become greatly affected, as is our ability to use water for recreation. To combat water pollution, we must understand the problem and become part of the solution.

Surface water pollutants are generally classified into these eight major categories:

1. sewage and waste
2. infectious agents
3. particulate soil and mineral matter
4. dissolved toxic pollutants.
5. mineral and chemical compounds
6. radioactive nuclides
7. thermal pollutants
8. organic chemical exotics

Surface water should be free from the following contaminants (1):

1. Compounds that impart color, turbidity, and odor such as greases, oils, toxic metals, phenols, and organics
2. Toxic radionuclides that acutely affect the physiology of humans, animals, and aquatic organisms
3. Substances that may precipitate to form objectionable deposits or float on the surface as oil, scum, or debris
4. Heavy materials that check the growth of aquatic flora and fauna
5. Thermal effluents should not increase the water temperature by 3 to 5 °F.
6. Substances that are likely to enhance the growth of undesirable aquatic life

Components of surface water are listed in Table 1.

Table 1. Typical Analysis of Some Surface Waters^a

Component (ppm)	X ^b	Y ^c
Silica	9.5	1.2
Calcium	4.0	36.0
Magnesium	1.1	8.1
Iron (III)	0.07	0.02
Total hardness as CaCO ₃	14.6	123
Sodium	2.6	6.5
Potassium	0.6	1.2
Sulphate	1.6	22
Chloride	2.0	13
Nitrate	0.41	0.1
Bicarbonate	18.3	119
Total dissolved solids	34	165

^aReference 2.

^bX: Pardee Reservoir, Oakland, California, U.S.A., 1976 (water supply source for East Bay Area of San Francisco).

^cY: Niagara River, New York, U.S.A. (water supply source for city of Niagara Falls).

SOURCES OF SURFACE WATER POLLUTION

Surface water comes in direct contact with the atmosphere, streams, rivulets, and surface drains. So a continuous exchange of dissolved and atmospheric gases occurs and wastes are added through water conveyance.

There are many causes of surface water pollution but two general categories exist: direct and indirect contaminant sources. Direct sources include effluent outfalls from factories, refineries, waste treatment plants, decomposed plant animal matter, and radioactive materials and wastes. Indirect sources include effluent contaminants that enter the surface water supply from surface soils and from the atmosphere via rainwater. Soils contain the residue of human agricultural practices (fertilizers, pesticides, etc.) and improperly disposed of industrial wastes. Atmospheric contaminants are also derived from human practices (such as gaseous emissions from automobiles, factories, and even bakeries).

Impact of Livestock Production Practices on Surface Water Quality

Livestock practices that can impact surface water quality include both intensive and nonintensive operations. Intensive operations include feedlots (>500 head of cattle), dairies, and wintering sites; nonintensive operations include pasture, cow-calf operations, and watering sites for cattle. Waste management and disposal can also impact water quality. Livestock density is not the only factor affecting water quality; siting and management are also important considerations. Water quality parameters related to livestock production include nutrients (nitrogen and phosphorus), microorganisms (e.g., bacteria, faecal coliforms, *Cryptosporidium*, *Giardia*), and organic materials such as livestock wastes.

Impact of Pesticides on Surface Water Quality

Pesticides can move into surface water by both point and nonpoint source pollution. Point source pollution occurs when pollutants originate from a single event or fixed site. Point source events include chemical runoff during improper storage, mixing/loading, disposal or misapplication to waterbodies. Nonpoint source pollution is the movement of pesticides from areas across watersheds over time into the surface water. Runoff (water flow) and erosion (soil particles) are common forms of nonpoint source pollution of surface water from agriculture. Pesticides are most susceptible to runoff immediately after application when they are in the "mixing zone." The mixing zone is the thin layer (1/10 to 1/3 inch) at the soil surface.

Impact of Fertilizers on Surface Water Quality

Fertilizers given to crops are not always fully consumed; part remains in the soil absorbed by soil colloids and influences the quality of surface water when it is dissolved. Chemical fertilizers consist mainly of relatively simple compounds of nitrogen, phosphorus, and potassium, which are nutritive elements for plants. Nitrogen when that enters surface water tends to cause eutrophication of lakes and other surface water reservoirs.

Impact of Mining

In many countries, water pollution related to the mining industry can be traced back to older times compared with that caused by other industries. The characteristics of waste water from mining industries vary greatly depending on the kind of mine (3).

Metal Mining Industry. Many kinds of wastewater are discharged. For example, mine water, ore dressing water, wastewater from smelters, in general, contain metal ions, sulfuric acid and suspended solids at high concentrations.

Coal Mining Industry. Wastewater from coal mines consists mainly of mine water, coal dressing water, trace and major elements and contains a large quantity of suspended solids.

Sulfur Mining. In general, wastewater from sulfur mines contains large quantities of sulfuric acid and iron and has a low pH.

Stone-Quarrying and Clay-Supply Industries. These industries supply limestone, building stones, porcelain clay, and so on. Wastewater from the sites of these industries generally contains large amounts of suspended solids. Mining activities generate large quantities of solid wastes in most cases from large heaps. Such wastes not only pollute surface water by exuding toxic substances and also by rapid generation of suspended solids in surface water when the heap is washed away by a heavy rainfall or a flood.

Food Processing Industry

A common characteristic of wastewater from all types of food processing industries is the high organic content. Materials contained in wastewater can be divided roughly into carbohydrate, protein, fat, and mixtures of them. In many instances, wastewater contains oil, nitrogen, and phosphorus. In starch production, sugar refining, and brewing, the BOD level of waste water is especially high.

Paper and Pulp Industries

These are typical water consuming industries, and the levels of COD and suspended solids in wastewater are extremely high. A large quantity of waste paper fiber is also discharged as suspended solids together with wastewater.

Iron and Steel Industry

Wastewater from the cooling and cleaning processes for coke furnace gas contains ammonia, cyanide, and phenols. There is also wastewater from the dust collecting process of each furnace that contains suspended solids (coke dust and ore) and that from the pickling process, which contains acid, iron, and oil.

Electroplating Industry

Toxic substances released through wastewater are various heavy metals (cadmium, zinc, copper, etc.), cyanide, hexavalent chromium, acids, and alkalis.

Additional Forms of Water Pollution

Additional forms of water pollution exist in the forms of petroleum, radioactive substances, and heat. Petroleum often pollutes surface waterbodies in the form of oil from oil spills. Besides supertankers, offshore drilling operations contribute a large share of pollution. One estimate is that one ton of oil is spilled for every million tons of oil transported. This equals about 0.0001%. Radioactive pollution can be human-made or natural. Radioactive substances are produced as waste from nuclear power plants and from the industrial, medical, and scientific use of radioactive materials, or they can come from naturally occurring radioactive isotopes in water like radon. Specific forms of waste are from uranium and thorium mining and refining.

The last form of water pollution is heat. Heat is a pollutant because increased temperatures result in the deaths of many aquatic organisms. These increases in temperature are caused by discharges of cooling water from factories and power plants.

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POND AQUACULTURE—MODELING AND DECISION SUPPORT SYSTEMS

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SOLAR RADIATION AND TEMPERATURE IN SHALLOW STAGNANT WATERS

Physical environment plays a controlling role in all processes and events that occur in waters. Knowledge of water's dynamic characteristics and model elaboration is basic. Shallow stagnant waters are characterized by the individuality resulting from their small capacity. This is displayed in their receptivity to external factors, such as weather variability, that differentiates them from lakes. In a temperate climate, quick and distinct changes in meteorological events are displayed in variations in the solar radiation that penetrates into shallow waters, thus, in variations in light and thermal conditions. These changes are then transferred directly and/or via chemical processes to different levels of the biological chain. In fishponds, the final link in the chain is the stenothermic organism which is affected by the ambient temperature in a complex way. Penetrating light plays the decisive role in fish survival, limiting the concentration of oxygen dissolved in the water. The thermal conditions inhibit

development and growth of the warm water fish, such as carp, throughout the considerable part of the year.

An attempt to approach the time-vertical distribution of penetrating solar radiation and temperature is presented in descriptive form and in simple mathematical language. The majority of the material is obtained from constructed carp ponds situated in a temperate climate ($\phi = 49^{\circ}52'N$, $\lambda = 18^{\circ}48'E$). Both the differentiated pond morphometry and the various periods of their use allow following their effects on the physical environment. Their average surface area is equal to some thousand square meters, and the average depth varies from 120 cm in the warm season to 200 cm in winter. The simultaneous time-vertical distribution of temperature is presented in ponds whose surface areas are from some hundred to some thousand square meters and from 20–120 cm deep (Fig. 1). The gravitational water flow through the ponds requires inclination of their bottoms from the water inlet to its outlet. In deeper ponds, the difference in depth between shallow and deep places can exceed 1 m. Dikes one to two meters high above the water surface limit the wind stress, especially in smaller ponds.

Two periods limited by water temperature are distinguished; warm—from the beginning of April to the end of October, when the water temperature is above $4^{\circ}C$, and cold—from November till March, when the temperature drops below $4^{\circ}C$. Most of the examples presented deal with period from May till September, when the thermal conditions stimulate growth and development of the carp organism.

In the warm part of the year the water feeding ponds compensates only for evaporation and infiltration losses. During the winter, the scanty water flow through ponds covered with ice protects them from oxygen deficiency. The solar radiation flux density was monitored in the wavelength range $0.3\text{--}3.0\text{ }\mu\text{m}$.

The Warm Season

Absorbed solar radiation is the main source of heat in shallow waters; the greatest heat losses take place in the evaporation process (1). In early spring, solar radiation



Figure 1. View from the air over the ponds of the Institute of Ichthyobiology and Aquaculture of the Polish Academy of Sciences at Golysz.

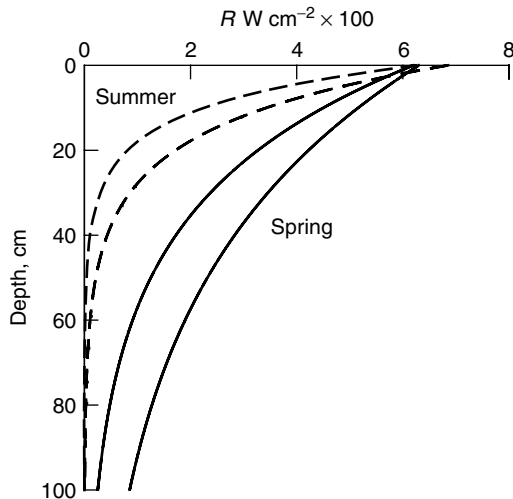


Figure 2. Density of the vertical solar radiation flux R under the free water surface in ponds during sunny and calm days; higher extinction presents ponds with a denser carp stock.

penetrates to the pond bottom. At night, cooling of the surface layers makes the whole water stratum circulate; a breeze enhances this process. Favorable light conditions during the day and nutrients released from the bottom during water circulation stimulate the development of flora and fauna and distinctly diminish the radiative flux penetrating to the bottom. Intensive absorption of the long waves and intensive scattering of the short waves of the radiative spectrum lead to high extinction in the surface layers and to an exponential decrease of R with the pond depth z (Fig. 2). On sunny days under the free water surface, $R(z)$ takes the form of

$$R(z) = R_0 e^{-\alpha z}$$

The coefficient α of vertical extinction is equal to 0.02 cm^{-1} in spring and increases in late summer; it also increases in ponds densely stocked with carp (Fig. 2) (2,3).

Values of α in ponds are approximately ten times higher than in most lakes, where they usually are computed per 1 m (4–6). A wind whose speed exceeds 3 m s^{-1} stimulates the circulation of the entire water stratum and also limits the penetration of vertical radiation to a certain degree. This distinct stratification of vertical radiation manifests in a distinct vertical decrease in temperature and oxygen content in ponds (Fig. 3).

In late summer in the deeper parts of ponds fertilized or densely stocked with carp, the oxygen concentration decreases below the saturation point (7). Macrophytes limit the radiation penetrating into ponds; however, the largest decrease occurs below dense floating plants. During sunny, calm days, the regular diel and seasonal solar radiative cycles follow the Sun's elevation in the whole water stratum (Fig. 4).

They can be approximately represented at all the depths by the geometric function

$$R(z, t)_d = \{A_R(z, t) \sin[\pi/T_R(z, t)]\}_d$$

Amplitude $A_{R,d}$ and period $T_{R,d}$ decrease approximately exponentially with depth. A decrease in $T_{R,d}$ shortens the day at the bottom (8). The seasonal cycle of $R(z, t)_s$ follows the Sun's elevation only in the surface water layer (Fig. 5). In the deeper parts, the average monthly radiative flux R decreases from May to September owing to decreasing water transparency.

An equation for the distribution $R(z, t)_s$ of seasonal-vertical solar radiation in ponds during sunny day is not a simple geometric function, though it can be satisfactorily computed using the first terms of the Fourier series

$$R(z, t)_s = \{\Sigma_n A(z) \sin[n\pi/T(z)t + \varphi(z)]\}_s$$

where $(A, T, \varphi)_s$ are the seasonal amplitude, period, and phase of $R(z, t)_s$ in the period from May–September.

The greater the surface area of ponds, the greater the role of wind in water circulation and in radiation extinction (9). Momentary cloudiness deforms the time-vertical

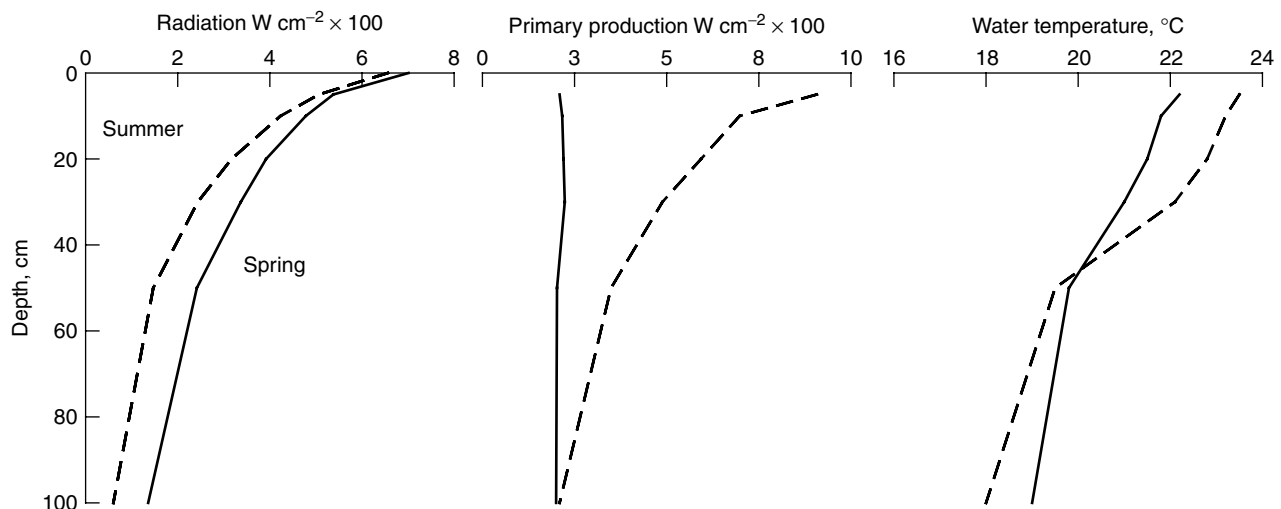


Figure 3. Example of the simultaneous vertical distribution of radiative flux, primary production, and water temperature in ponds on a sunny, calm summer day.

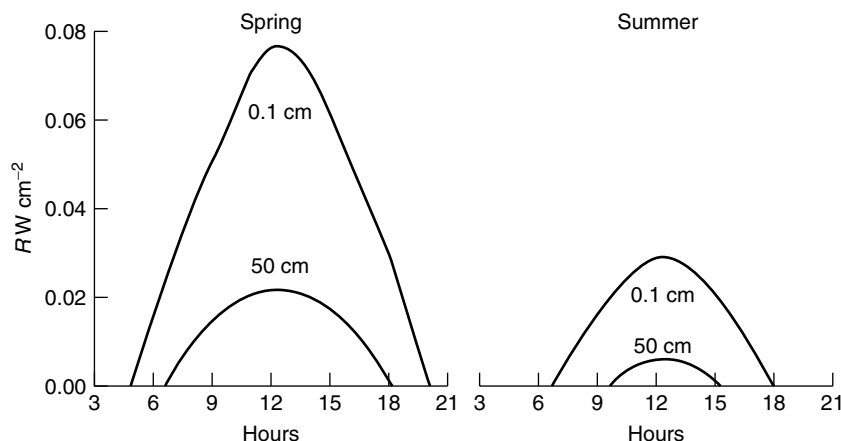


Figure 4. Diel cycles of radiative flux in ponds on sunny days.

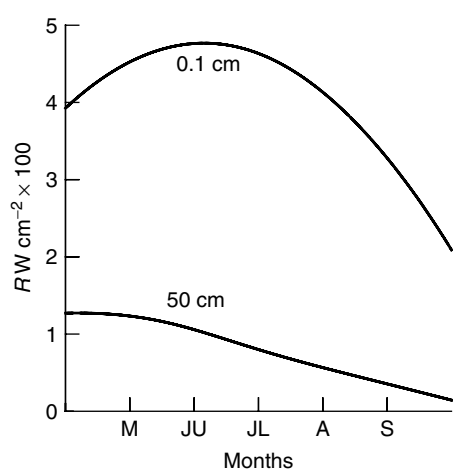


Figure 5. Seasonal cycles of radiative flux in ponds on sunny days.

radiative flux and diminishes its density; cloudiness maintained for a whole day causes the daily radiative amplitude to approximate zero.

In shallow waters, the time-vertical distribution of the temperature ϑ is chiefly a resultant value of the absorbed radiation and heat losses in the evaporation process. During sunny, calm days, temperature cycles are semiregular geometric functions; that have with a slightly quicker increase in temperature during the morning hours and a distinctly slower decrease in the afternoon (Figs. 6 and 7). During the night, the temperature difference between surface and bottom water layers approaches zero. The diel amplitude and vertical gradient decrease exponentially as the average depth z of the pond increases (10,11).

In May, when sunny but cold northern air masses advect, the temperature in very shallow waters, for example, about 20 cm deep, can decrease to a few degrees in the early morning and increase to over 25°C at noon. These great time variations may be lethal for some organisms, such as e.g., carp fry (12).

The Cold Part of the Year

In late autumn and in winter, the radiative flux reaches only a few percent of that recorded in summer and

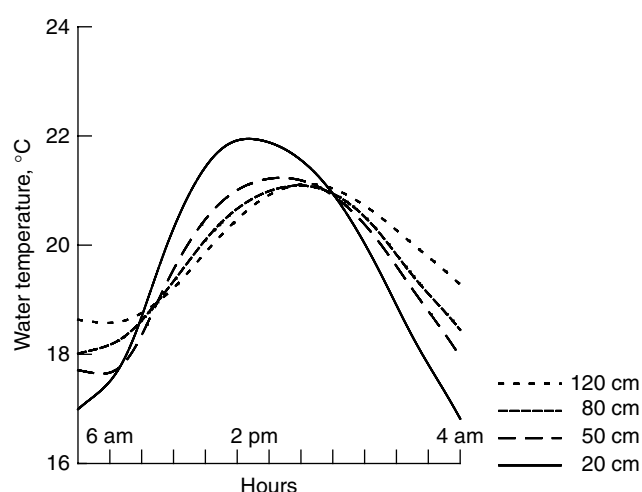


Figure 6. Simultaneous smoothed diel temperature cycles in ponds of different average depths on a sunny, calm summer day.

is additionally limited in ponds covered by ice. Only transparent ice allows radiation to penetrate to the bottom and to create favorable conditions for primary production. Under nontransparent ice, water flow through ponds is required to improve oxygen conditions. In non frozen ponds, the action of wind in stirring water ensures sufficient oxygen concentration for the fish.

The first autumnal deep cooling equalizes and lowers the temperature in the whole water stratum to 4°C. A further temperature decrease is manifested in inverse stratification. Strong wind accompanied by cooling stirs the water, obstructs freezing, and lowers the temperatures in the whole stratum, sometimes even below 0.1°C. In ponds covered by ice, the temperature in the lowest water layers is heated from the bottom and usually increases to 2–2.5°C. Such vertical temperature distribution is normally maintained from the end of December to early March. The warm, windy weather in the second half of this month removes the ice cover and raises the water temperature to 4°C, equalizing differences between the surface and the bottom (10).

In a temperate climate, the ice cover usually disappears one or more times from ponds (13), at first leading to momentary water heating and, under the consecutive

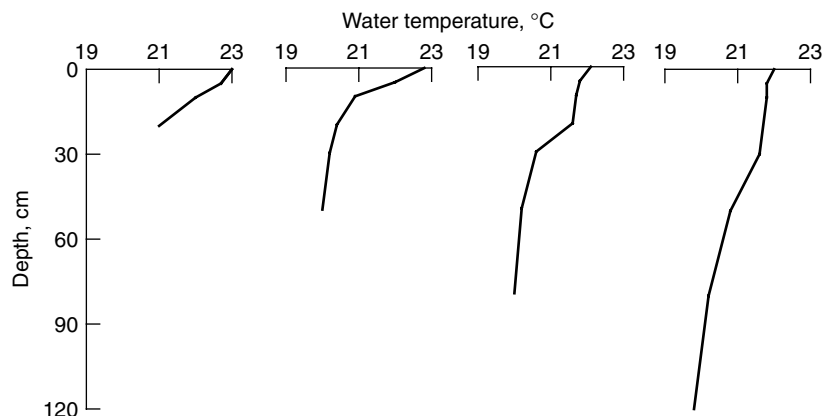


Figure 7. Simultaneous vertical temperature distribution in ponds of different average depths on a sunny, calm summer day.

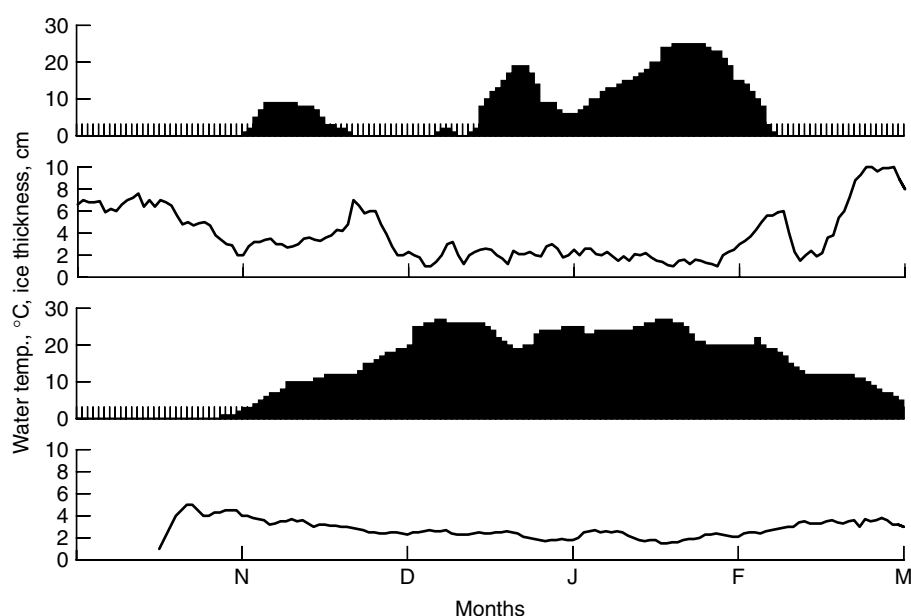


Figure 8. Examples of the average diel temperature patterns in the pond's bottom water layer during winter for momentary disappearances and permanent ice cover.

cooling, to a deep temperature decrease (Fig. 8). This creates unfavorable conditions for warm water organisms, for example, it increases the mortality of carp juveniles.

SUMMARY

The individuality of shallow stagnant waters, such as carp ponds, results from their small capacity and differentiates of their physical environment from lake environments. The density of the solar radiative flux that penetrates into water and the vertical temperature and oxygen stratification in ponds situated in a temperate climate are characterized by high extinction and vertical gradients, approximately ten times higher than in most lakes. Vertical extinction and attenuation increase from spring to late summer. Despite its high variability, average diel radiative courses can be represented by a simple geometric function, the seasonal distribution by the first several terms of the Fourier series. The diel cycle of water temperature in ponds corresponds to the annual cycle in lakes. The effect of increasing pond depths is displayed in the exponential decrease of the diel temperature

amplitude and vertical gradient. During winter in ponds covered by ice, the temperature in the bottom water layer is maintained between 2 and 2.5 °C. Short warm weather periods lead to the momentary disappearance of ice and to a temperature increase, the consecutive cooling—to a considerable temperature decrease.

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PUMPING STATIONS

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Surface water pumping stations are employed either to move water from an available surface location to where it is wanted or to remove it from a location where it is not wanted. Examples of the former include municipal or industrial water supply and irrigation, whereas municipal drainage pumps are examples of the latter.

Although differing in purpose, all have similarities of function and components. A pumping station must provide a water inlet for water to flow to the pump, a pump suitable for the water and delivery requirements, and a means of powering the pump. Although these requirements are interdependent to some degree, a reasonable approach is to consider them in order.

WATER INLET

Surface water frequently carries with it flotsam, jetsam, and aquatic species, both plant and animal, which must be prevented from entering the pump without severely restricting the flow of water. In addition, surface water levels often fluctuate. Settling basins, reservoirs, grids, screens, and multiple intakes or floating intakes have been used to address these problems.

Reservoirs and settling basins even out the water supply reducing fluctuation in the water level and because of reduced velocity of water flow, allow debris either to settle out or float to the surface. The reduced water flow also reduces the impact from objects in the water, which reduces the required strength of the intake structure.

It is preferable to obtain the water from a high-quality spot. For example, in many reservoirs, a preferred area is about 1 meter below the surface. As the level of the reservoirs change, this entails for a fixed structure, multiple ports located at different levels and valved so that the desired level may be chosen. Low entering velocities are desirable to avoid heavy loading of floating matter, sediment, and fish. Velocities of less than 0.5 ft/sec are preferable. Higher velocities result in heavier loading and greater maintenance requirements of the intakes. Current and foreseeable environmental conditions should be considered when siting the intakes. For example, winds that stir up sediments near shore may necessitate siting the intake away from the shoreline, as may pollution caused by shoreline development. Submerged intakes are possible if they do not interfere with navigation. Some small stations may employ floating intakes in which the intake is mounted on a floating structure such as a raft and connected to the pump with flexible pipe.

Pumps are guarded by physical barriers such as inlet grids or screens located upstream of the pump. These are usually arranged in series, beginning with those that have the largest openings and, by necessity, are the largest and strongest and proceeding to more restrictive barriers. These barriers must be cleaned periodically to prevent the accumulation of debris from severely restricting the water flow. Cleaning may be manual or automatic.

PUMPS

A pump (or pumps) is the second major component of a pumping plant. A pump must be capable of producing the required head while delivering the desired flow rate. It must also be able to handle the suction lift and the water quality or lack thereof. The latter is dependent on the water—its corrosiveness (salty water or chemical laden) and how much and what kind of trash or debris it may contain. An appropriate pump is constructed of materials that resist the corrosive characteristics of the water and of a design that is capable of handling the anticipated debris.

The head, suction lift, and flow rate are dependent on the piping system and the pump's characteristics. The piping system and pump interact to determine the operating point of the pump—flow rate and pressure. The pump cannot independently control these parameters. As the flow rate increases, the head (pressure) required to force the water through the pumping system also increases. A pump will typically have reduced capacity (flow) as the head it is pumping against increases. Thus, as the flow rate increases, the pressure a pump is capable of producing decreases while the pressure required to move the water increases. The operating point occurs at that flow rate under which the pressure produced by the pump is equal to the pressure required to move the water. The efficiency of a pump varies greatly with its operating

point, with maximum efficiency occurring at a particular flow rate and falling off at higher or lower flow rates.

Manufacturers typically list an operating range in which the pumps performance is satisfactory; outside this range, the efficiency of the pump may be low and damage to the pump could occur. The required output of a pumping plant often varies over time. In order to avoid operating outside of the pumps designed range, a pumping station may employ multiple pumps or, when feasible, storage. Multiple pumps, although more costly initially, allow the flexibility of operating the pumps at an efficient operating point and also supply redundancy in case of pump failure.

Water is stored to equalize pumping rates over time. One example of storage is a system of drainage canals that themselves are drained by a pump (as in New Orleans, where much of the city lies below sea level and water must be pumped to provide drainage). The drainage canals provide storage volume as well as a means of flowing the water to the pumping station. Economies result from operating smaller pumps at their rated loads rather than large pumps at conditions less than optimum for the pump.

Two types of pumps are typically used in pumping stations—centrifugal and axial flow. Centrifugal pumps use centrifugal force to move water from one point to another and to develop pressure. Water enters at the center of an impeller, where it is spun and forced at a high velocity to the outer edge by the rotating impeller. The water is discharged into a casing where some of the velocity is converted into pressure. In an axial flow pump, both the water flow and the force exerted on the water by the pump is in a straight line along the axis of the pump. Axial flow pumps use a propeller or screw to apply force to the water. Axial flow pumps are particularly suited for high-volume low-head (pressure) applications. Their efficiency is high, especially when the head is in the range of 8 to 20 feet. An advantage of an axial flow pump is its ability to handle some debris. Centrifugal pumps, particularly high efficiency ones, often have close tolerances and may not handle trash as well. Centrifugal pumps are capable of operating efficiently at higher heads than axial flow pumps.

Suction lift is suction that a pump must apply to suck water into its inlet. It includes the height the pump is located above the water source, the friction loss in the pipe leading from the water source to the pump, and the energy imparted to the water as velocity. The amount of suction lift that a pump can produce is limited and varies with the individual pump. Excessive suction lift can cause inefficient operation, damage to the pump, or a failure to operate. For these reasons, it is preferable for the pump to be immersed in the water source or for the water to flow to the pump under gravity.

POWER

Pumps are powered by electric motors or internal combustion engines. Internal combustion engines include diesel, gasoline, natural gas, and liquefied petroleum gas (LPG) engines, with diesel being the most common. Each power source has advantages and disadvantages, many of which are site- and application-dependent. Deciding

factors may include ability to do the job, reliability and availability of the power source and fuel supply, cost of equipment and installation, expected useful life, cost and ease of maintenance, convenience of operation, and the cost of energy to operate.

With the large variety of motors and engines available, a suitable example of either should be available and capable of powering the chosen pump. In general, internal combustion engines become more applicable as the power requirements and hours of use increase. Fuel efficiency of higher horsepower engines is usually better than for low horsepower engines (when each is properly matched to the load), and the higher fixed costs of engines as compared with motors can be spread over more operating hours as the hours of use increase. In addition, engines may be operated at varying speeds that provide more flexibility than most electric motors.

Advantages to electric motors exist if the cost of providing service and the cost of electricity, including energy, demand, and standby charges, are not prohibitive. The electric motor provides ease of operation, long life, requires minimal maintenance, and maintains its performance level year after year. In addition, initial costs are usually less than the cost of internal combustion engines, and the reliability is higher. They can, however, be shut down by the loss of electrical power. If the ability to operate during times when power is frequently lost (for example, during periods of flooding after a hurricane) is a necessity, or the pumping station is located in an area that suffers from frequent power outages, the internal combustion engine may be a better choice.

Natural gas engines and electric motors are dependent on the availability of natural gas service and electric service, respectively. If this service is not available, or the cost of obtaining it is too great, than diesel, gasoline, or LPG engines, which can be supplied with fuel from storage tanks, become the feasible power sources. Storage tanks allow considerable freedom in siting the power unit, but they also present additional concerns. The tanks should be designed to prevent pollution caused by leaks, spills, or evaporation, which includes guarding against leaks or spills and provides a means of containing the spill and of cleaning after a spill. For this reason, underground tanks are usually avoided. Above-ground tanks have their own sets of problems. They are more vulnerable to attack or vandalism and tend to have a greater degree of fuel evaporation than underground tanks. Not only does evaporation result in fuel loss, it may also lead to higher gum content of the fuel and poor engine performance and clogged filters, which, if severe enough, can prevent an engine from operating. Adulteration of the fuel can occur by the condensation of water inside the tank or, in the case of diesel fuel, by bacteria that feed on the fuel.

Energy costs are estimated under the conditions at which the pumping station operates. A thorough analysis requires the expected operating schedule including loading, time of occurrence, and duration of operation. Loading is particularly important in determining internal combustion engine fuel usage as the efficiency of delivered power is greatly impacted by the degree to which the engine is loaded. For example, Dr. Claude Boyd of Auburn

University obtained comparative efficiency data of an engine under different loading conditions. The engine, running at 1800 rpm, required 1.6 gal/hr of fuel to provide 4.9 horsepower. Increasing the load to 16.9 horsepower, a greater than three-fold increase, required 2.0 gal/hr, or only 25% greater fuel consumption. Similarly, the same engine running at 950 rpm required 0.7 gal/hr to supply 4.8 horsepower and 1.2 gal/hr to supply 16.7 horsepower. Under these different loading conditions, the number of horsepower hours produced by the same engine for each gallon of fuel consumed ranged from 3.1 to 13.9. Although not generally as sensitive to loading as internal combustion engines, electric motors also fall off in efficiency when underloaded, particularly at 50% or less of capacity. Energy costs for electricity is impacted by operating schedule as electric charges for commercial customers include demand and service charges in addition to energy costs. Demand and energy charges are charges for having electric power available, even if it is not used.

Maintenance costs can significantly affect the viability of a particular power option. Maintenance costs are considerably higher for internal combustion engines than for electric motors. Exact costs are difficult to pin down as they depend on duty cycle, cost of labor, degree of maintenance, and the individual engine or motor. A rough rule of thumb is to access internal combustion engines a 1 to 1.5 cent per horsepower-hour maintenance cost penalty in comparison with electric motors.

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REGULATED RIVERS

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The regulation of river flows by dams and reservoirs has been fundamental to the advancement of civilization, and as dryland development continues, river regulation for water supply and power production continues to be central to economic growth. However, river ecosystems have evolved to become adapted to the natural pattern of flow variability from season to season and from year to year. Flow regulation has had dramatic impacts on both running-water and floodplain ecosystems, resulting in major changes in biodiversity and to fisheries.

River regulation involving the redistribution of water in space and time has been fundamental to the advancement

of human society (see Water in History). Dams have been built to impound rivers and store runoff in large reservoirs for use in dry seasons, in drought years, for long-distance transfer to centers of high water demand—for irrigation agriculture or urban use—and for hydroelectric power production. It has been estimated that 800,000 dams of all sizes have been built on the world's rivers. The largest have dams over 150 m high and reservoirs with storage capacities greater than 25,000 million cubic meters or have a hydroelectric generation capacity of more than 1000 MW. Today, large dams contribute directly to about 15% of global food production and nearly 20% of the world's electricity supply (1). However, these dams, reservoirs, and diversions have had widespread ecological impacts. Worldwide, reservoirs have drowned about 1 million km² of land and in the United States 25,000 km of fast-flowing main rivers have been converted into stillwater lakes and on the Columbia-Snake system alone, 822 km of river habitats have been replaced by reservoirs. Here, the focus is on the regulated rivers downstream from dams.

By 1994, it was estimated that nearly 80% of the large river systems in Europe, the former Soviet Union, the United States, and Canada were at least moderately regulated by dams, reservoirs, and diversions (2). In the United States, the nation's major rivers, the Mississippi, Colorado, and Columbia, are completely regulated by a combination of large dams and weirs. Impoundments trap a river's sediment load and about 25% of the global flux of river sediments is trapped in reservoirs. Reservoirs can also alter the water quality of the river downstream of the dam, especially the annual pattern of temperature variations. However, the major impact of dams and reservoirs is to regulate the flow regime along the river below the dam and in many cases also to significantly reduce the total runoff (3).

In the 1990s, worldwide annual water withdrawals of 4430 km³/yr represented between 35% and 50% of the average annual renewable resource, and irrigation agriculture consumed some 2500 km³ of water. Obvious indicators of unsustainable water use include the failure of the Hwang He and Colorado Rivers to reach the sea in dry seasons and the shrinkage of the Aral Sea by 25,000 km². In the case of the Nile River, the influence of the Aswan Dam on the flow regime extends for 1000 km, changing the hydrographic conditions over the continental shelf of the southeast Mediterranean. Ecological indicators include the loss of productive floodplain wetlands, dramatic changes of channel size and shape, the degradation of river and floodplain fisheries, and the incursion of saltwater into coastal floodplain and delta areas.

Flow regulation includes the reduction of average annual runoff; reduced seasonal flow variability or reversal of the seasonal pattern to meet irrigation demands; altered timing of annual extremes, reduced flood magnitudes, and the imposition of unnatural, short-duration pulses associated with hydropower generation. Until the 1960s, most dams and reservoirs were designed and operated with little regard for the river and its flow-dependent ecosystems. In most cases, the only discharges from the dam were compensation flows—often a constant minimum flow—to meet the needs of industry along the river,

especially for water mills, and other riparian uses, with higher flows during the rare periods of dam spill.

During low flow periods, many regulated rivers became dry until unregulated tributaries restored some flow. A survey in 1987 by the Institute of Hydrology of supply, regulating, and hydropower reservoirs in the United Kingdom showed mean compensation flows to the river below the dam of 15–17% of average daily flow, but for reservoirs operated only for water supply the compensation flow was zero. In dryland areas, often minimum flows were increased during the natural low-flow season to meet the needs of irrigation and other abstractors downstream, and along many rivers minimum water levels have been raised by weirs to meet navigation requirements.

All reservoirs regulate floods. The basic concept of flood regulation is empty space and reservoirs having a large flood-storage capacity in relation to the annual runoff can exert complete control upon the annual hydrograph of the river downstream. However, even when a reservoir is full to spillweir level, temporary storage of floodwaters within the lake plays an important role in reducing the maximum rate of outflow from the reservoir when the water area of a reservoir is 2% or upward of the catchment area. Along many regulated rivers, the mean annual flood has been reduced by more than 25% and there are examples where the 50-year flood has been reduced by over 20%.

Flood regulation and the maintenance or increase of minimum flows has led to major changes in the size and shape of river channels (4). Degradation has occurred immediately below some dams because of the trapping of the sediment load behind the dam, but the reduction of channel size in response to the reduced flood magnitudes has been dramatic—channel capacities being reduced by 50% or more in some cases. Rates of channel narrowing have been particularly rapid along some semiarid rivers where the maintenance of perennial flows encouraged the rapid establishment and growth of woody vegetation on exposed sediments. Elsewhere, rapid aggradation has occurred at confluences between regulated rivers and tributaries carrying high sediment loads forming deltas and bars that can extend for several kilometers below the confluence.

In the past, the most disturbing feature of regulated rivers was that the environmental changes resulting from regulation were not anticipated. A scientific journal, *Regulated Rivers*, was founded in 1988 to provide a focus for synthesizing studies of the environmental impacts of river regulation. In 2002 the journal changed its name and emphasis to *River Research and Applications*, reflecting major changes in both research on, and management of, regulated rivers. For 25 years attention had been moving to the determination of flows required to protect water-dependent ecosystems. By the start of this millennium there had developed a vanguard movement to restore regulated rivers and to reconnect rivers with their floodplains; to advance decision-making processes to balance consumptive water use with that needed to maintain a healthy river; and to allocating our limited water resources to meet the needs of both humans and

riverine ecosystems (5). This issue is addressed in this volume under Environmental Flows.

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RESERVOIRS-MULTIPURPOSE

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NEED FOR RESERVOIRS

Reservoirs are developed for two main functions. The first is to store water to even out the fluctuations in river flow and match the availability to demand. Generally, the major part of annual stream flow is available during the few months of the rainy season, but the demand for water arises all year round. Therefore, it is necessary to store the excess water in the months of high flows and use it when the natural stream flow is inadequate to meet the demand. The water stored in a reservoir may also be diverted to faraway places where it is needed.

The second function is to raise the level of the water upstream of the dam to provide a hydraulic head. The creation of storage and head allows dams to generate electricity. The other important purpose of a reservoir is to control flooding in downstream areas.

The early Mesopotamians were the first dam builders. The Sumerians built networks of irrigation canals in the plains along the lower Tigris and Euphrates Rivers about 6500 years ago. Earth and rock-filled dams had been built around the Mediterranean, in the Middle East, China, and Central America by the late first millennium B.C. The remains of impressive dams and aqueducts in Spain are fine examples of the ingenuity of Roman engineers. South Asia has a long history of dam building. Long earthen embankments were built to store water for irrigation. The remains of the Indus Valley civilization which flourished 4000 to 5000 years ago show that these people had built well-planned networks of water supply and drainage works. The oldest reservoir in operation today is the Aftatang reservoir (capacity 100 million m³), constructed west of Shanghai during 589 to 581 B.C. (1). The oldest continuously operating dam still in use is the Kofini flood control diversion dam and channel constructed in 1260

B.C. on the Lakissa River upstream of the town of Tiryns, Greece, which it continues to protect.

Worldwide, there were more than 36,000 large dams by 1986 (2); many more have been constructed since then. The Asian continent accounts for more than 64% of all dams, and China has built most of them. Engineers of the former USSR have built most of the large reservoirs; they are followed by engineers from Canada. More than 80% of the dams in the world are earth and rock-filled types.

A reservoir contains a number of structural features other than the main dam. Spillways are used to discharge water when the reservoir level threatens to become dangerously high. Dams built across broad plains may include long lengths of ancillary dams and dikes. A schematic diagram of a reservoir is given in Fig. 1.

A reservoir project requires huge amounts of money, manpower, land, and other resources and significantly affects the environment, population, and economy of the region. Once these projects are in place, it is not easy to undo their impacts. Therefore, they should be carefully planned to impart the maximum possible benefit to the national economy.

Multipurpose Reservoirs

The typical purposes of a reservoir are water supply for domestic and industrial uses, irrigation, navigation, generation of hydroelectric power, recreation, and flood control. Depending on the purposes served by the reservoir, it may be classified as either single purpose or multipurpose. A single purpose reservoir serves only one purpose. A multipurpose reservoir serves a combination of purposes. As a rule of thumb, the bigger a reservoir, the more purposes it can serve.

While planning a reservoir, all available data should be analyzed, and, if necessary, further information should be gathered so that the best decisions are made with respect to location, size and type of structure, and auxiliary facilities. After the best site is selected from amongst the potential sites, additional investigations are carried out to finalize the project details. These include engineering, geologic and hydrologic investigations, and economic analysis. The water yield at a site is commonly estimated using flow duration curves.

Storage Zones in a Reservoir

For ease of analysis and operation, the entire reservoir storage space is conceptually divided into a number of

zones by drawing imaginary horizontal planes at various elevations. The lowest zone is the dead (or inactive) zone. The bulk of the storage capacity for conservation is provided in the conservation (or active) storage zone. The top level of the conservation zone is termed the full reservoir level (FRL) or normal pool level. If the storage space above the FRL is exclusively reserved for flood control, the maximum storage capacity is dead plus active storage. The maximum water level (MWL) is the highest elevation to which the water is allowed to rise in a reservoir.

Dead storage is provided in a reservoir for accumulating sediments or to provide the minimum head for the turbines of a hydropower dam.

Reservoir Storage–Yield Analysis

The procedures estimating the storage capacity needed to meet given demands or the possible yield from a given project constitute the storage–yield analysis. The reservoir storage–reliability–yield relationship relates inflow characteristics, reservoir capacity, release, and reliability. The inflow process, the reservoir storage, and the outflow process constitute the stream-flow regulation system. Its components are the reservoir storage capacity (S), yield (q), and a measure of reliability (R). The relationship among these variables can be symbolically designated by a function (3):

$$\phi = \phi(S, q, R) \quad (1)$$

where $S > 0$, $q > 0$, $0 < R \leq 100\%$. This function is called the regulation regime function, or storage–yield function.

Critical period techniques are the earliest methods of storage–yield analysis. The critical period is the period in which an initially full reservoir, passing through various states (without spilling), empties. The *mass curve method* proposed by Rippl was the first rational method for computing the required storage capacity of a reservoir. This simple method is commonly used in the planning stage. Define a function $X(t)$ as

$$X(t) = \sum x(t) dt \quad (2)$$

where $x(t)$ represents inflows. The graph of $X(t)$ versus time is known as the mass curve. The reservoir capacity is obtained by finding the maximum difference between cumulative inflows and cumulative releases for the most critical period of recorded flow. This method, although

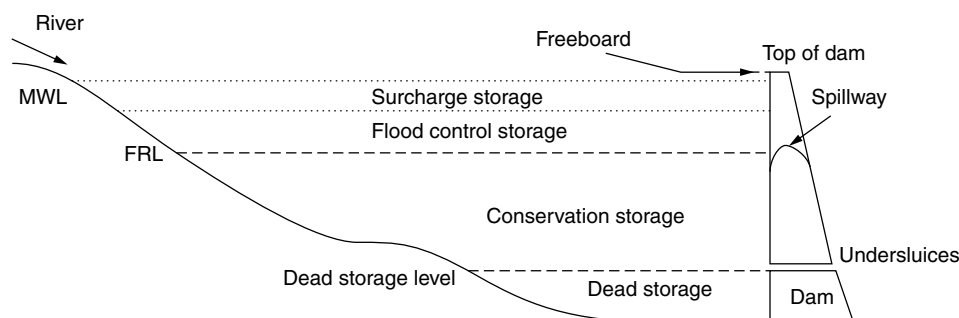


Figure 1. Schematic diagram of a reservoir.

simple and straightforward, has a few shortcomings. It is suitable when the draft is constant, and it is not possible to consider evaporation losses properly. The sequent peak algorithm overcomes some of the limitations of the mass curve method.

Systems analysis techniques such as optimization and simulation are frequently used for storage–yield analysis. Among the optimization techniques, linear programming and dynamic programming have been particularly suitable. Simulation is a widely used technique for solving a variety of problems in designing and operating a water resources system. The reason is that this approach can be realistically and conveniently used to examine and evaluate the performance of a set of alternative options. The simulation approach can be used as a stand-alone method or can be used to modify further and test the results of critical period or optimization methods. McMahon and Mein (4) describe storage–yield analysis techniques in detail.

Flood Control Storage Capacity

The requirement of storage for flood control conflicts with the requirements for conservation. Conservation requirements, such as water supply and hydropower generation, require the storage space to be full, whereas flood control requires empty storage space.

The demands for water supply and hydroelectric power are relatively deterministic, whereas the demand for flood control storage is highly stochastic. Furthermore, the usual time period for analysis of conservation is 1 month, whereas for flood control, it is of the order of a few hours. The storage requirement can be estimated by using the design flood hydrograph. An initial reservoir level at which this flood hydrograph impinges on the reservoir is assumed. The maximum level attained by the reservoir is computed by routing the hydrograph through the reservoir. The maximum height of a dam is obtained after adding the freeboard to this level.

RESERVOIR OPERATION

The efficient use of water resources requires judicious design and also proper operation. After a reservoir comes into being, the benefits that could be reaped depend largely on how well it is operated. Reservoir operation forms an important part of planning and managing a water resources system. Postconstruction, detailed guidelines are given to the operator which enable him to make appropriate management decisions. A reservoir operating policy specifies the amount of water to be released from storage at any time depending on the state of the reservoir, demands, and any information about the likely inflow to the reservoir. The operating problem for a single-purpose reservoir is to decide about the releases to be made, so that the benefits for that purpose are maximized. For a multipurpose reservoir, additionally, it is required to allocate the release optimally among various purposes. The complexity of the problem of reservoir operation depends on the extent to which the various intended purposes are compatible. If the purposes are compatible, comparatively less effort is needed for coordination.

Conflicts in a Multipurpose Reservoir

While operating a multipurpose reservoir, a number of conflicts can arise in satisfying various purposes. These are as follows:

Conflicts in Reservoir Space. These conflicts occur when a reservoir (of limited storage) is required to satisfy incompatible purposes, for example, water conservation and flood control. The satisfaction of conservation purposes requires the reservoir to be filled to the maximum extent, whereas the objective of flood control is best met when the reservoir has sufficient vacant space. The critical decision in regulation is whether to fill the reservoir or keep it vacant.

Conflicts among Purposes. Conflicts can also arise among conservation purposes when the pattern of water use is different and the requirement of storage space for one purpose is not in conformity with the other. The water required for uses such as irrigation, municipal, and industrial water is consumed and cannot be shared with any other use.

Conflicts within the Same Purpose. A deficit of water in a reservoir can be distributed over time in different ways. A typical decision is whether to cut the supply now so that there is a small deficit for a longer period or to postpone the cut for the future and risk a bigger shortage for a shorter time. The impact of these two decisions will be different.

Simple Approaches to Reservoir Operation

A reservoir is operated according to a set of rules to store and release water, depending on the purposes it is required to serve. The release decisions are made in accordance with the available water, current and forecasted inflows, demands, weather outlook, and time of the year. The drawdown–refill cycle of a reservoir is usually 12 months long, except when the reservoir capacity is large in relation to stream flows. In many regions of the world, the filling and drawdown periods are distinctly separated.

Standard Linear Operating Policy

The simplest of reservoir operating policies is the standard linear operating policy (SLOP), graphically represented in Fig. 2. According to this policy, if the amount of water available in a particular period, is less than the target demand (T), all available water is released. If the available water is more than the target demand but less than target demand plus available storage capacity, a release equal to the target demand is made, and the excess water is stored in the reservoir. If there is no space to store the excess water even after meeting the target demands, all the water in excess of the maximum storage capacity is released.

Note that SLOP is a one-time operating policy; future implications of the current decision are ignored. This type of time-isolated release of water is neither beneficial nor desirable. Though this policy is not used in day-to-day operation, it is frequently used in planning studies.

Rule Curves

A rule curve or level specifies the desired storage to be maintained in a reservoir during different times of

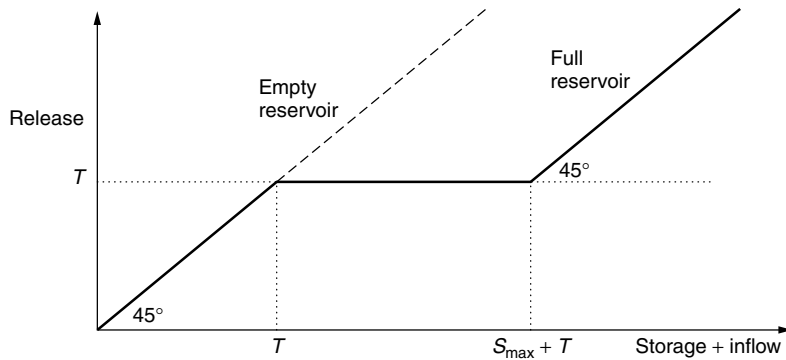


Figure 2. Graphical representation of standard linear operating policy.

the year while trying to meet various demands. The implicit assumption is that a reservoir can best satisfy its purposes if the storage levels specified by the rule curve are maintained at different times. Different rule curves may be developed for different purposes, such as water supply for domestic and industrial use, irrigation, hydropower generation, and flood control. The rule curve as such does not give the releases from the reservoir; these depend on actual inflows and demands.

The derivation of rule curves depends on the type and purposes of the reservoir. The data from the critical period are used for this purpose. Clearly, if the reservoir can meet the demands during the critical year, it will be able to serve its purpose for all other years. These days, many software packages are available for preparing rule curves.

While operating a reservoir following rule curves, if the reservoir level at any time is above the rule curve, releases are made to meet all conservation demands. If the reservoir water level is in the vicinity of the rule curve level, the release is restricted so that the reservoir level does not fall appreciably below the rule curve. If for some reason, the reservoir level drops below the rule curve, the release should be curtailed with the objective of returning to the rule curve at the earliest.

Rule curves implicitly reflect the established trade-off among various project objectives in the long run. These curves specify only the ideal levels to be maintained; the operators can use their experience and judgment to distribute the excess or deficit to maximize benefits. The operation of a reservoir may become quite rigid by strictly following the rule curves. To provide flexibility, different rule curves may be specified for different circumstances,

viz., normal, dry, and wet. An operating schedule that gives some leverage to the operator to use his experience and hydrologic forecasts is termed a flexible schedule. Of course, this requires using detailed input data, good models, trained manpower, and careful planning. But at the same time, the benefits can be very large.

Concept of Storage Zoning

The entire reservoir storage space can be conceptually divided into a number of zones by drawing imaginary horizontal planes at various elevations. The sizes of these zones can vary with time as shown in Fig. 3.

Reservoir operators are expected to maintain the reservoir level within the specified zones. This conceptual division of a reservoir into a number of zones and the rules governing the maintenance of reservoir levels in a specified zone are based on the assumption that, at a specified time, an ideal storage zone exists for the reservoir and benefits can be maximized by keeping the storage in this zone. This concept is in some way akin to a rule curve and has the added advantage that it gives more flexibility to the decision-maker.

Operation of a Multireservoir System

The benefits from the joint operation of a system of reservoirs can be substantially larger than the sum of benefits from the operation of individual reservoirs. A system may consist of reservoirs in series, in parallel, or a combination. Various approaches for developing operating policies for a system of reservoirs have been discussed by Wurbs (5).

While managing multiple reservoirs, all reservoirs are maintained in the same zone at any time, to the

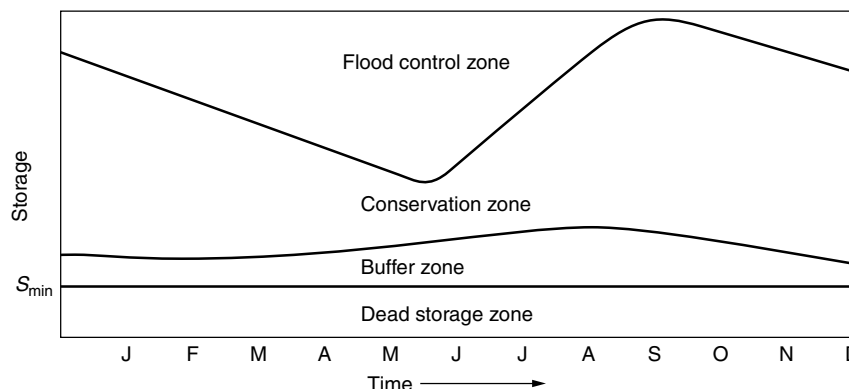


Figure 3. Variation of reservoir zones with time.

extent possible. This is necessary to have balance among reservoirs after an unexpected or extreme hydrologic event. There are three approaches for balancing reservoir contents. The first, known as the "equal function" policy, is to keep all reservoirs at their same zonal position. The second is based on a reservoir ranking or priority concept—the entire zone of the lowest ranking reservoir is used fully before drawing water from the next lowest ranking reservoir, and so on. The third concept is based on a "storage lag" policy. Withdrawals from the zones of some reservoirs are made before withdrawals are begun from the same zones of other reservoirs. After a certain volume has been released from the initial group of reservoirs, releases are made from all reservoirs, maintaining the percentage difference of the available zone volumes.

Conditional rules are also used to regulate multiple-reservoir systems. These policies define reservoir releases as a function of the existing storage volumes and the time of the year and also as a function of the expected natural inflows into the reservoirs for some prespecified future time period.

Reservoir Operation for Flood Control

A storage reservoir with gates is one of the most effective means of flood control. The moderation of a flood through storage is achieved by storing part of the flood volume in the rising phase of the hydrograph and releasing it gradually in the receding phase of the flood. The degree of moderation or flood attenuation depends on the empty storage space available in the reservoir when the flood impinges it. The flood control pool must be emptied as quickly as downstream flooding conditions allow; this will reduce the risk of highly damaging future releases, should major floods occur in quick succession.

In multipurpose reservoirs located in the regions where floods can be experienced at any time and flood control is one of the main purposes, space is permanently allocated for flood control at the top of the conservation pool. The amount of space depends upon the magnitude of likely floods. If the floods are experienced only in a particular season, space is allocated only during that season.

Reservoir regulation consists of storing the peak flows over and above the safe (nondamaging) carrying capacity of the channel at the damage point in the reservoir. The reservoir is emptied after the passage of the flood to make space for controlling subsequent floods. In Fig. 4, ABCDE

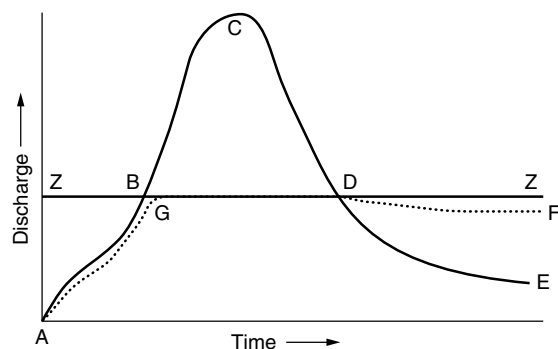


Figure 4. Ideal operation of a reservoir for flood control.

represents the inflow hydrograph. The line ZZ represents the nondamaging carrying capacity of the river channel downstream of the reservoir. From point B to point D, the natural flow in the river exceeds its safe carrying capacity. If there were no reservoir, the flood water would have spilled over the channel banks and caused damage during the time B to D. The moderated release from the reservoir under ideal operation is given by the dotted curve AGDF. As soon as the inflow begins to increase, the release is gradually increased, till some point G where the release equals the safe carrying capacity. While the inflows from point B to D exceed the safe carrying capacity of the downstream channel, the release is maintained within a safe range by storing the volume in the segment BCD in the flood control zone of the reservoir. After point D, the inflows continue to fall rapidly but the release, while still in a safe zone, exceeds the inflow, so that the reservoir is quickly emptied.

This is an ideal situation which is possible only if the perfect foreknowledge of the hydrograph is available. In actual operation, the release curve will deviate from the ideal shape. For example, if the operator makes smaller releases in the early part of the hydrograph, it is likely that the reservoir will completely fill before point D. In that eventuality, the operator will be forced to make releases at rates exceeding the safe carrying capacity of the downstream channel thereby causing flood damage. Conversely, if at the beginning of a flood, the operator starts making higher releases in the expectation of a major flood and such a flood does not occur, the reservoir may not fill to the desired level by the end of the filling season.

SYSTEMS ENGINEERING FOR RESERVOIR MANAGEMENT

Determination of the reservoir operating policy for efficient management of available water is a complex problem. Many attempts have been made to solve this problem using optimization and simulation models.

Optimization is the science of choosing the best solution from a number of possible alternatives. Optimization methods find a set of decision variables such that the objective function is optimized. Linear programming (LP) and dynamic programming (DP) techniques have been extensively used in water resources. Loucks et al. (6) illustrated applications of LP, nonlinear programming (NLP), and DP to water resources. Yeh (7) presented a state-of-the-art review and discussed in detail the various optimization models for multipurpose, multireservoir operating problems. Simonovic (8) provided a state-of-the-art review and applications of the systems approach to reservoir management and operation. Wurbs (9) reviewed the application of optimization, simulation, and network-flow models.

Simulation is the process of designing a model of a system and conducting experiments with it to understand the behavior of the system and to evaluate various strategies for operating it. The essence of simulation is to reproduce the behavior of the system. It allows controlled experimentation without disturbing the real system. However, simulation analysis does not yield an immediate optimal answer which can be arrived at iteratively. The simulation technique is possibly the most powerful tool for studying complex systems.

The simulation models associated with reservoir operation include the mass-balance computation of reservoir inflows, outflows, and changes in storage. They may also provide an economic evaluation of flood damage, hydropower benefits, irrigation benefits, and other similar characteristics. A large number of reservoir planning and operating studies have used simulation models because such models can provide a more realistic representation of reservoir systems and their operations. They also allow added flexibility to derive responses which cannot be readily defined in economic terms (recreational benefits, preservation of fish and wildlife). The results of simulation models are easier to explain to decision-makers and system operators because the ideas inherent in simulation modeling can be understood easily. Thus, these models are effective tools for a dialogue with operators and decision-makers.

A number of general-purpose computer software programs are available nowadays that can be used for analysis related to planning, design, and operation of multipurpose reservoirs. Some well-known simulation models are the HEC-5 model developed by the Hydrologic Engineering Center, (10), the Acres model (11); the Streamflow Synthesis and Reservoir Regulation (SSARR) Model (12), and the Interactive River System Simulation (IRIS) model (13). Jain and Goel (14) presented a generalized simulation model for conservation operation of a multipurpose reservoir system. Lund and Guzman (15) discussed derivation of operating rules for reservoir systems.

Real-Time Reservoir Operation

A multipurpose reservoir can be efficiently operated if the time interval between the occurrence of an event and the execution of the control adapted for that event is short. In real-time operation, release decisions for a finite future time horizon are made based on the condition of the reservoir at the instant when these decisions are made and the forecast about the likely inflows/demands over this time horizon. After a certain time interval, the new information about the system becomes available, the forecasts are updated, and the decisions are modified.

Real-time operation is especially suitable during floods where the catchment response changes rapidly and decisions have to be made quickly and adapted frequently. Successful application of the real-time operating procedure requires a good telemetry system through which data can be observed on-line. Real-time flood forecasting involves estimating discharge in a river at some period prior to its occurrence. It is very useful in real-time operation because the forecast leadtime proves useful in mitigating some of the adverse effects of flooding. Real-time hydrologic forecasting depends, to a large extent, on timely availability of good quality hydro-meteorological data at a forecasting station.

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DAM REMOVAL AS RIVER RESTORATION

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River is a dynamic system. It makes spatial and temporal adjustments of its channel geometry, profile, and pattern to maintain a dynamic equilibrium condition compatible with changing independent variables or constraints applied to the river system. The dynamic adjustments of a river are accomplished through the processes of scour, sediment transport, and deposition. Human activities, especially the construction and operation of dams, have profoundly affected most rivers in the world. As a result of these activities, many rivers have lost their ability to maintain or recover their dynamic equilibrium and functions at a self-sustained level. The main purpose of river restoration is to reestablish the basic functions of transporting water and sediment to sustain the ecosystem of rivers and their corridors.

Dams are built and operated for flood control, irrigation, water supply, hydropower generation, navigation, recreation, and environmental purposes. Each dam was designed to last for a given useful lifetime with certain economic and other benefits in mind. A dam can also impose certain risk to downstream residents and properties. Most of the dams, especially small dams, were designed with an estimated useful life of 50 to 100 years. A dam may be removed because of one or more than one of the following factors:

- Economics
- Dam safety and security
- Legal and financial liability
- Ecosystem restoration
- Fish passage improvement
- Site restoration
- Recreation

Sediments are trapped in reservoirs after the completion of dams. The amount of sediments trapped in a reservoir depends on the trap efficiency of the reservoir, which is a function of sediment inflow, ratio between average annual inflow and reservoir volume, reservoir shape, and the dam's operational rules and facilities. A detailed sediment management plan must be developed before a dam's removal to prevent:

- Excessive downstream river channel aggradation, channel widening, bank erosion, increase of flood stage, plug water intake structure, turbidity, and descript aquatic habitats
- Adverse impact on water quality
- Excess sediment removal cost and difficulties of finding appropriate sediment storage site
- Head cutting of tributaries

Engineering considerations of dam removal include complete or partial removal; materials used in the dam construction; rate of dam removal and reservoir drawdown; and ability to draw reservoir pool and construction of bypass channel or low-level outlets. Table 1 summarizes some of the factors that need to be considered in dam removal.

Dam removal can be an engineering method for river restoration. Because of the dynamic nature of river adjustments in response to natural and manmade events, comprehensive studies of possible responses of a river must be carried out before a dam's removal. Lane (1) proposed the following qualitative equation to maintain a dynamic equilibrium among sediment load Q_s , sediment size d , water discharge Q , and river slope S .

$$Q_s d \propto QS \quad (1)$$

Table 1. Relationship Between Dam Decommissioning and Sediment Management Alternatives [modified from ASCE (4) by Randle and Greimann (5)]

Sediment Management Alternative	Dam Decommissioning Alternatives		
	Continued Operation	Partial Dam Removal	Full Dam Removal
No Action	• Reservoir sedimentation continues at existing rates, • Inflowing sediment loads are reduced through watershed conservation practices, or • Reservoir operations are modified to reduce sediment trap efficiency.	• Only applicable if most of the dam is left in place. • The reservoir sediment trap efficiency would be reduced. • Some sediment may be eroded from the reservoir.	• Not applicable.
River Erosion	• Sluice gates are installed or modified to flush sediment from the reservoir. • Reservoir drawdown to help flush sediment.	• Partial erosion of sediment from the reservoir into the downstream river channel. • Potential erosion of the remaining sediment by sluicing and reservoir drawdown.	• Erosion of sediment from the reservoir into the downstream river channel. Erosion rates depend on the rate of dam removal and reservoir inflow. The amount of erosion depends on the ratio of reservoir width to river width.
Mechanical Removal	• Sediment removed from shallow depths by dredging or by conventional excavation after reservoir drawdown.	• Sediment removed from shallow depths before reservoir drawdown. • Sediment removed from deeper depths during reservoir drawdown.	• Sediment removed from shallow depths before reservoir drawdown. • Sediment removed from deeper depths during reservoir drawdown.
Stabilization	• The sediments are already stable because of the presence of the dam and reservoir.	• Retain the lower portion of the dam to prevent the release of coarse sediments or retain most of the dam's length across the valley to help stabilize sediments along the reservoir margins. • Construction of a river channel through or around the reservoir sediments.	• Construction of a river channel through or around the existing reservoir sediments. • Relocate a portion of the sediments to areas within the reservoir area that will not be subject to high-velocity river flow.

where Q and Q_s = water and sediment discharge, respectively; d = sediment particle diameter, and S = channel slope.

Yang (2) introduced the following quantitative equation for the prediction of dynamic adjustment of a river based on his unit stream power equations (3,4)

$$C_t = I \left(\frac{VS}{\omega} \right)^J \quad (2)$$

or

$$\frac{Q_t}{Q} = I \left(\frac{QS}{WD\omega} \right)^J \quad (3)$$

where W = channel width, D = channel depth, ω = sediment fall velocity, C_t = total bed-material concentration, Q_t = total bed-material load, and I, J = coefficients.

Because sediment fall velocity is directly proportional to the square root of sediment diameter and J has an average value of 1.0 for rivers, (3) can be simplified to

$$\frac{Q_t d^{0.5}}{K} = \frac{Q^2 S}{WD} = \frac{Q^2 S}{A} \quad (4)$$

where A = channel cross-sectional area, d = median sediment partial diameter, and K = a site-specific parameter.

Equation (4) was used by Yang (7) for the prediction of river morphologic changes because of the construction and operation of a dam. Equation (4) can also be used for the prediction of dynamic adjustments of a river after the removal of a dam.

The rate and timing of a staged reservoir drawdown during the dam removal process should follow the following criteria, especially for the purpose of river restoration (6):

- The drawdown rate is slow enough so downstream flood wave does not cause damages.
- The release of coarse sediment is slow enough not to cause excessive downstream aggradation.
- The concentration of fine sediment is not too great or duration too long to cause an unacceptable impact on water quality and aquatic environment.

During the dam removal process, a comprehensive monitoring program should be carried out to determine

- Reservoir sediment erosion and redistribution
- Hill slope stability along reservoir and downstream river channel adjustment
- Water quality and suspended sediment concentration
- River bed aggradation and flood stage
- Aquifer characteristics
- River channel plan form and channel geometry adjustments
- Effect of large woody debris
- Effects on coastal process

Sediments in a reservoir can be removed by dry excavation, mechanical dredging, hydraulic dredging, sediment conveyance by river flow, and partial sediment removal and reservoir stabilization. Sediment removal by river flow is

the least lost and most suitable method for river restoration purposes. This method also requires detailed studies of the hydrologic, hydraulic, sediment transport, scour, deposition, and river morphology. Because of the complexity of these studies, computer models are often used by experienced engineers to simulate and predict the dynamic adjustment of a river after the dam removal.

The U.S. Army Corps of Engineers' HEC-6 (8) and Bureau of Reclamation's GSTARS3 (9) and GSTAR-1D (10) are some of the commonly used computer models in public domain. Bureau of Reclamation's models and user's manuals can be downloaded by users by going to <http://www.usbr.gov/pmts/sediment/> and following the links therein. In addition to computer models, physical models are also used for short-term simulation and prediction of erosion, sediment transport, and deposition during the dam removal process.

Dam removal and river restoration often require a multidisciplinary team approach with team members consist of engineers, environmentalists, river morphologists, fishery and aquatic experts, and representatives from local community. Useful references include, but are not limited to, books on *Stream Corridor Restoration: Principles, Process, and Practice* (11), *Erosion and Sedimentation Manual* (12), and *Sediment Transport: Theory and Practice* (7).

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RIPARIAN SYSTEMS

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Riparian areas are those along the bank of a watercourse, which is a relatively narrow transition zone between a downslope aquatic waterbody and an upslope terrestrial watershed. Riparian areas are predominantly associated with a river or stream, but also define the bank of a lake, wetland, or other standing water. The riparian system refers to the atmospheric, vegetation, and soil components; their physical, chemical, and biological properties; and their interaction with each other and the upslope terrestrial and downslope aquatic hydrologic cycle. It is important to recognize that riparian areas may border both influent and effluent streams, known as those that lose or gain water to the groundwater.

DEFINITION

Riparian systems may be wetlands, but there are nonwetland riparian areas, and there is no jurisdictional definition of a riparian system as there is for wetlands, defined under Section 404 of the Clean Water Act (33 CFR 328.3(b)). For a discussion of the wetland function of riparian areas, see the entry for WETLANDS. As for wetland areas, various federal agencies have attempted to define riparian systems to help in identify them. According to the U.S. Fish and Wildlife Service (FWS), either the presence of a set of obligate water-loving vegetative species (e.g., phreatophytes that have water-table roots) different from that of adjacent areas or more rigorously growing vegetative species similar to upslope areas distinguish the riparian area. The U.S. Department of Agriculture (USDA) considers that riparian areas are further distinguished from the surrounding lands by saturated soils resulting from a shallower depth to the water table, which implies a distinct hydrologic regime.

SYSTEM INPUTS, OUTPUTS, AND TRANSFORMATIONS

A riparian system, in hydroecological terms, can be viewed from its inputs, outputs, and transformations. Inputs arrive via the atmosphere, surface soil, and subsurface soil media, and outputs may travel within these same media after hydrologic and biochemical fluxes between and within the atmospheric, vegetative, and soil components. Atmospheric inputs include short-wave, or solar, and long-wave radiation, wet and dry precipitation, and wind-advected constituents. Surface inputs include upslope storm runoff water, overbank water from a flooded watercourse, eroded soil, as well as sorbed, entrained, and dissolved chemical species. Subsurface inputs include local and regional discharge from upgradient groundwater flow paths, bank storage from the watercourse, and dissolved and entrained chemical species.

Riparian systems are open systems that provide complex and useful chemical sinks, sources, and internal

transformations of constituents important in aquatic chemistry and watercourse health. Energy flow into the riparian system, which provides riparian biota growth, reproduction, and maintenance, is derived principally from high-energy sunlight and the use of its short-wave radiation in autotrophic photosynthesis to form organic material from water and carbon dioxide. In the presence of an electron donor, such as oxygen, the organic material is harvested for its energy by heterotrophic organisms in respiration, the reverse process of photosynthesis. Because this energy is associated with the organic material, the food web energy flow has an associated mass cycle. In this cycle, the carbon and associated nutrients (e.g., phosphorous and nitrogen) pass through numerous organisms up the food web, and after excretion or organism death, the carbon and nutrients stop moving up the food web and are either transformed or decomposed back to their inorganic forms as part of mineralization. Microbes, such as bacteria and fungi, and rotifers (e.g., worms) are the key agents in transformation and decomposition.

Riparian input constituents include nutrient species (e.g., nitrogen and phosphorous), chemical pollutants from agrochemicals (e.g., herbicides and pesticides), metals from urban and industrial activities (e.g., lead, zinc, and copper), organic materials from watershed activities (e.g., waste from livestock or domestic pets and vegetative debris), suspended solids (e.g., sand, silt, and clays from eroded soils), pathogens from wildlife (e.g., *Giardia* and *Cryptosporidium*), and in residential areas, thermally heated water and human trash.

In time, the riparian system may accumulate these constituents from surface water that slows and deposits entrained constituents, from soil adsorbing (at the surface) and absorbing (internal to the structure) dissolved or entrained constituents, from vegetation taking the constituents into its root or stem system, or from water stores that fill in the surface or subsurface zone. Biochemical processes operating in the riparian system may transform material inputs that are retained, and another component of inputs that exceed storage may then discharge as outputs with subsequent wind advection, seasonal and storm based surface runoff, seasonal watertable fluctuations, and subsurface flow.

Riparian atmospheric and surface component transformations include mainly photolysis, saprophytic decay, animal digestion, and weathering. Microbial (e.g., fungi and bacteria) subsurface transformations are likely to fluctuate between those of aerobic and anaerobic conditions as the water table fluctuates. Fungi are solely aerobic, but bacteria can operate in both aerobic conditions and anaerobic and have a greater range of transformative options. Two governing principles for microbial transformations of chemicals require that they be energetically favorable and that enzymes are available as catalysts.

The biodegradation potential for a chemical is determined by the free energy change in its reaction with other environmentally available chemicals. This is determined from knowledge of the reduction–oxidation, or redox,

potential of the riparian area, which has an ecological sequence from aerobic (O_2) to denitrification (NO_3^-), to manganese reduction (MnO_2), to iron [$Fe(OH)_3$] reduction, to sulfate reduction (SO_4^{2-}), to methanogenesis (CO_2). Given that many enzymes are extremely specific and that mutations occur in bacteria, some individuals in a given strain of bacteria may accomplish transformations not observed in other individuals.

VEGETATED RIPARIAN BUFFERS

Riparian zones supporting forest or grass cover are described structurally as vegetative filter strips (VFS) or functionally as riparian buffer zones (RBZs). Three defined zones for the riparian buffer have been identified by the USDA (1) for agricultural nonpoint source (NPS) runoff and water quality management. The most upslope area is zone 3 and as proposed, contains grasses that slow NPS runoff velocities and allow settling out of entrained constituents such as sediment and sorbed pollutants. The middle area, zone 2, dedicated to managing forestland that captures overland flow leaving zone 1, provides a root-based carbon substrate for microbes (fungi and bacteria) transforming nutrients in subsurface flow, as well as income for the landowner. The near-stream area, is zone 1, which is an undisturbed forest, provides function and habitat for the aquatic system. There are no fixed recommendations for the width of these zones, and many landowners are cautious about dedicating otherwise revenue generating land for forested buffers. In some states, recommended widths for the entire buffer have averaged between 10 and 30 m.

In zone 1, direct biotic and abiotic controls of the vegetative buffer on the aquatic system are many. The vegetative canopy provides cover from incoming solar radiation and blocks outgoing evening long-wave radiation for stable temperature control, as well as protection from predators. The vegetative zone also drops large wood debris into the watercourse for in-stream habitat; creates turbulent eddies (e.g., hydraulic circular flows) and conditions favorable for mobile species seeking to feed or rest by creating areas of refuge from strong downstream currents; and provides substrate for aquatic plants and sessile organisms. Further, the vegetative root wads and downed stems along the bank provide soil stability and erosion control, and the small woody and leaf debris enter the aquatic food web and feed shredders, nymphs; algae, and trout.

Zone 3 and 2 overland flow interception, slowing, settling, and filtering of runoff-entrained constituents requires that the runoff enter in a thin, dispersed sheet and that concentrated channels of flow be removed and flow respread. Methods for spreading the flow include constructed and natural methods (e.g., limbs from a tree). In addition to surface channels of short-circuited flow discharging directly to the watercourse and thereby avoiding riparian storage and transformation, flow can also preferentially short-circuit in the subsurface soil. Such short-circuiting might include preferential flow through macropores and agricultural tile drains that pass too quickly for effective microbial transformation,

as well as deep groundwater flows that upwell into the stream at its base and bypass the microbial populations (see Fig. 1). Additional management interventions include removing accumulated sediment berms from the upslope edge of zone 3, as needed, and replacing vegetative materials that are not fixing or assimilating nutrients at target efficiency.

Vegetative riparian structure and function for the adjacent watercourse, it has been widely documented, provide critical resources for aquatic ecosystem health. Inside the United States, a great deal of this work was performed in the Coastal and Piedmont Regions in Maryland, Virginia, and North Carolina; studies in southern Finland and Sweden are also widely cited (2). Most studies have focused on nutrient transformations of nitrogen and phosphorous, and results indicate that nutrient retention varies as a function of nutrient load, riparian upslope steepness, and riparian buffer width, as well as vegetative types, health, subsurface flow paths, and microtopography (e.g., surface short-circuiting). Total nitrogen reduction estimates given for subsurface inputs are between 67 and 89% (3). Phosphorous reduction in U.S. riparian buffers ranges from 41 to 93% (4), and removal efficiency, it has been observed, decreases with time.

RIPARIAN BUFFER BEST MANAGEMENT PRACTICES

In U.S. Environmental Protection Agency (USEPA) research in the mid-Atlantic highland areas of Pennsylvania, Maryland, and Virginia, random sampling networks of benthic macroinvertebrate health, recorded as a biotic index (BI), trend strongly with riparian habitat health. For watersheds with land use stress, such as agricultural or residential occupancy, stream reaches that have low BI values (e.g., impaired) were also areas where riparian habitat was typically absent or degraded, whereas reaches that have high BI values had attendant healthy riparian habitat. Recommendations from these studies include using forested riparian areas as watershed restoration and best management practices (BMPs) to reduce upslope pollutant loads and improve aquatic health.

Hydrologic investigations in urban environments, such as the National Science Foundation (NSF) funded Long Term Ecological Research (LTER) research site in Baltimore, Maryland, have found differences between urban and rural riparian systems. Although the riparian

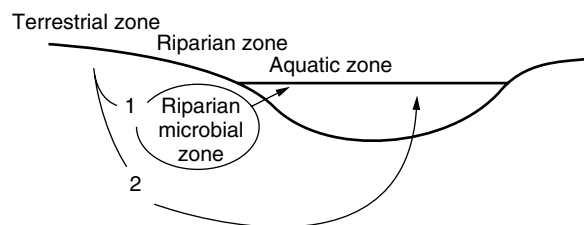


Figure 1. Flow paths from terrestrial to the aquatic zone showing (1) passage through the riparian microbial zone and (2) bypass beneath the riparian microbial zone where nutrient transformations occur.

system in both urban and rural cases is the transition between terrestrial and aquatic processes, the gradual blending of these zones found in rural areas is more abrupt in urban areas. In fact, in areas where streams have been channelized for flood control, leveed from their natural floodplain, and upslope ground water recharge has been reduced by impervious cover, urban riparian systems are drier than their counterpart rural systems. In such cases, vegetative species that are obligatory water loving will not survive in the urban riparian system that has greater depth to the water table and more pronounced mesic periods. Instead, the more common species found in urban riparian systems are obligatory highland, or dry tolerant, trees and shrubs.

For new storm water management under the Clean Water Act, known as the Municipal Separate Storm Sewer System (MS4), the USEPA has required permitting for most municipalities to document pollutant discharge reductions. These reductions may be obtained by BMPs and for retrofitting to existing urban structure, will require smaller footprint design and engineering than a traditional 10 to 30 m wide riparian system. One BMP design that borrows water treatment functional components from riparian buffers is referred to as a vegetated bioretention device, or rain garden. In a USEPA Factsheet, this device is suggested for redirecting runoff from parking lots, driveways, and urban streets, when collocated with storm drain inlets, vertically through the BMP and into the groundwater or a subsurface drainpipe. The mixture of vegetation, soils, and microbes will transform and reduce total Kjeldahl nitrogen (TKN, e.g., ammonia and organic nitrogen) by 68 to 80%; total phosphorous by 70 to 83%; lead, copper, and zinc metals by 93 to 98%; and total suspended solids, organics, and bacteria by 90% (5). Given that this system is intended for urban areas where water tables are, deeper plant species should be drought tolerant, and in colder regions, salt tolerant in areas where roads are deiced.

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RIVERS

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Rivers are natural streams typically larger than a creek. The distinction may be somewhat subjective, but creeks cannot be traveled by boat and rivers can. Rivers are the result of precipitation and runoff over land surfaces, followed by surface erosion and sediment transport. As water runs off to lower elevations, the erosion of the underlying earth and soil carves channels into the landscape. These natural channels convey the floodwaters from rainfall and snowmelt and also drain the aquifers during periods of low flow. Rivers are perennial when there is flowing water all year round. Rivers that dry out during part of the year are ephemeral streams, also called arroyos. Related nouns include riverbank or riverside to describe the banks of a river, as well as riverbed to designate the stream bottom. Other adjectives related to rivers include riverine, riparian, fluvial, and fluvatile.

The discharge, or rate of outflow, of a river depends on the channel width, flow depth, and flow velocity. The flow velocity depends on the flow depth, or hydraulic radius, the slope of the river, and the roughness of the riverbed. The volume of water that flows through a river is affected by the duration and intensity of rainfall precipitation or snowmelt in the drainage basin of the river. Over geologic times, it is possible for rivers to be deflected into a different drainage system. This phenomenon is called a capture and results in an increase in drainage area.

Traditionally river systems have been classified as young, mature, or old depending on their stage of development over geologic times. A young river typically flows in a steep valley and has a steep bed slope and numerous riverbed irregularities. Cliffs or escarpments and differences in the resistance of rocks create irregularities in a riverbed and can thus cause rapids and waterfalls. Young systems are characterized by discontinuous longitudinal profiles, lakes, and variable hydraulic geometry. At a mature stage, the river valley has a wide floor and flaring sides and more pronounced erosion by its tributaries. An old river system tends to have regular longitudinal profiles and a gradual decrease in riverbed slope in the downstream direction. Geologic discontinuities such as faults and active tectonics can cause sudden changes in bed elevation that effectively can rejuvenate old fluvial systems. The age classification of rivers is diminishing in popularity now that quantitative studies of river behavior are more common.

The topography of natural rivers depends on the balance between the quantity of sediment in the stream and its transport capacity. A river is alluvial when it can transport its own riverbed sediment and can form its own channel geometry. The weathering process of the land of a given watershed produces large masses of alluvium that can be carried by rivers over geologic periods. An alluvial river tends to eliminate irregularities and form a smooth

Table 1. Water and Sediment Loads of Selected Rivers^a

River	Catchment Area 10 ⁶ km ²	Discharge			
		Water		Sediment	
		m ³ /s	mm/yr.	10 ⁶ ton/yr	mm/yr
Amazon	7.0	100,000	450	900	90
Mississippi	3.9	18,000	150	300	55
Congo	3.7	44,000	370	70	15
La Plata/Parana	3.0	19,000	200	90	20
Ob	3.0	12,000	130	16	4
Nile	2.9	3,000	30	80	15
Yenissei	2.6	17,000	210	11	3
Lena	2.4	16,000	210	12	4
Amur	2.1	11,000	160	52	15
Yangtse Kiang	1.8	22,000	390	500	200
Volga	1.5	8,400	180	25	10
Missouri	1.4	2,000	50	200	100
Zambesi	1.3	16,000	390	100	50
St. Lawrence	1.3	14,000	340	10	6
Niger	1.1	5,700	160	40	25
Murray-Darling	1.1	400	10	30	20
Ganges	1.0	14,000	440	1,500	1,000
Indus	0.96	6,400	210	400	300
Orinoco	0.95	25,000	830	90	65
Orange River	0.83	2,900	110	150	130
Danube	0.82	6,400	250	67	60
Mekong	0.80	15,000	590	80	70
Hwang Ho	0.77	4,000	160	1,900	1,750
Brahmaputra	0.64	19,000	940	730	800
Dnjepr	0.46	1,600	110	1.2	2
Irrawaddi	0.41	13,000	1,000	300	500
Rhine	0.36	2,200	190	0.72	1
Magdalena (Colombia)	0.28	7,000	790	220	550
Vistula (Poland)	0.19	1,000	160	1.5	5
Kura (USSR)	0.18	580	100	37	150
Chao Phya (Thailand)	0.16	960	190	11	50
Oder (Germany/Poland)	0.11	530	150	0.13	1
Rhone (France)	0.096	1,700	560	10	75
Po (Italy)	0.070	1,500	670	15	150
Ishikari (Japan)	0.016	230	450	6	270
Tiber (Italy)	0.013	420	1,000	1.8	100
Tone (Japan)	0.012	480	1,250	3	180
Waipapa (New Zealand)	0.0016	46	900	11	5,000

^aAfter Reference 1.

gradient from its source to its base level. The lowest level to which a river flows is called the base level. Sea level is the ultimate base level, but a lake or reservoir may serve as a local and temporary base level.

Slight changes in sediment transport capacity or in sediment supply to a stream can result in a significant change in bed elevation and planform geometry. For instance, if a stream receives more sediment than it can transport the result is aggradation, or a raise in riverbed elevation through time. Conversely, when the sediment transport capacity exceeds the upstream supply, the water will erode the riverbed and riverbanks and degrades the riverbed.

Similar changes normally take place laterally in river bends. Typically, the velocity increases on the concave riverbank and the increased sediment transport capacity results in erosion and lateral migration of a meandering channel. Simultaneously, the decreased flow velocity near

the convex bank of the river decreases transport capacity and results in sediment accumulation and aggradation through time to form point bars. As it approaches base level, downward cutting is replaced by lateral cutting, and the river widens its bed and valley and develops a sinuous course that forms exaggerated loops and bends called meanders. A river may open up a new channel across the arc of meander, called a neck cutoff, thereby cutting off a river segment and creating an oxbow lake as shown in Fig. 1.

River velocity is one of the primary factors that control the quantity and size of rock fragments and sediment carried by the river. When rivers reach oceans, lakes, or reservoirs, the decrease in flow velocity results in net deposition of the sediment load. Part of the load carried by the stream is deposited in the riverbed or on the floodplain beyond the channel. The flow of water during floods exceeds the conveyance of the stream channel, and

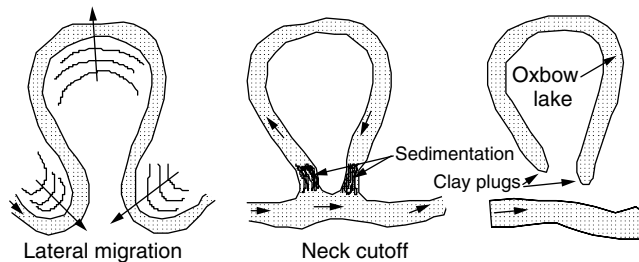


Figure 1. Oxbow lake formation process.

the water level increases to overtop the riverbank and flood adjacent floodplains. As the velocity reduces further on the floodplain, the deposition of sediment forms natural levees. Over long periods of time, landforms produced by deposition include floodplains, deltas, and alluvial fans.

Some typical characteristics including the watershed size, the water and sediment discharges of large rivers are presented in Table 1. The flow and drainage area of the Amazon far exceed that of any other river. On the other hand, the Yellow River (Huang Ho) and the Ganges have tremendous sediment concentrations for very large streams. Several textbooks describe the mechanics of river formation. Some of the recommended texts for future reading and reference in river morphology and river mechanics include Julien (1), Yalin and da Silva (2), Wohl (3), Yang (4) and the recent text of Bridge (5).

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RIVER AND WATER FACTS

National Wild and Scenic Rivers
System - National Park Service

WATER FACTS

- River Lengths
- The National Wild and Scenic Rivers System has only 11, 303 river miles in it—just over one-quarter of one percent of our rivers are protected through this designation, and this protection is often contended.

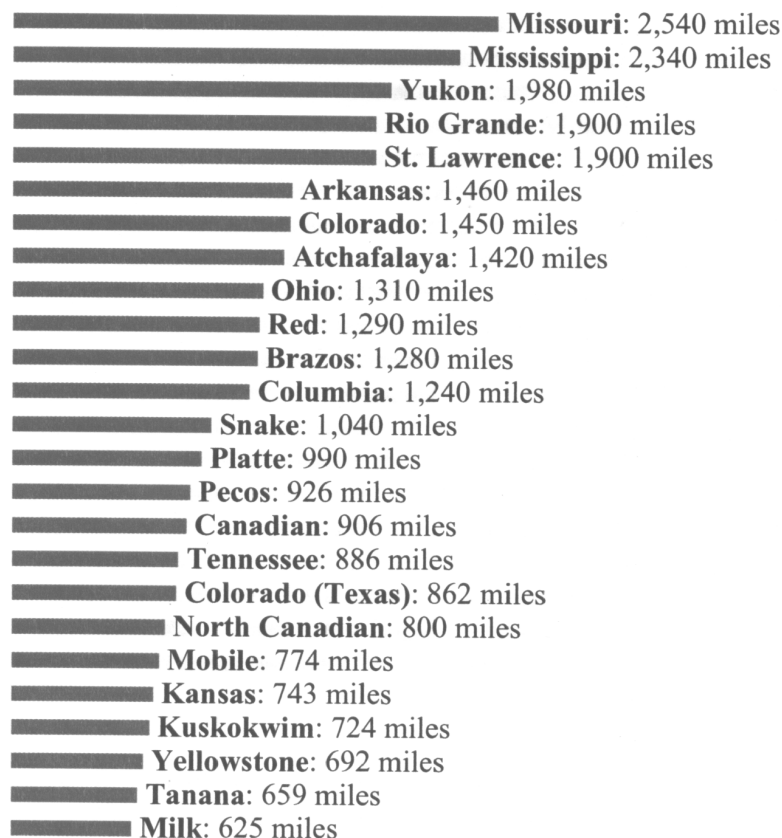
- Currently, 600,000 miles of our rivers lie behind an estimated 60,000 to 80,000 dams.
- The United States has 3,500,000 miles of rivers. The 600,000 miles of rivers lying behind dams amounts to fully 17% of our river mileage.
- The Missouri River is about 2,540 miles long, making it the longest river in North America. The Nile is the longest river in the world at 4,132 miles as it travels northward from its remote headwaters in Burundi to the Mediterranean Sea.
- The 8 longest rivers in the U.S. are (in descending order) Missouri, Mississippi, Yukon, St. Lawrence (if you count the Great Lakes and its headwaters as one system), Rio Grande, Arkansas, Colorado, Ohio.
- The 8 largest rivers in the U.S., based on volume, are (in descending order) Mississippi, St. Lawrence, Ohio, Columbia, Yukon, Missouri, Tennessee, Mobile.
- Water covers nearly three-fourths of the earth's surface.
- Most of the earth's surface water is permanently frozen or salty.
- Over 90% of the world's supply of fresh water is located in Antarctica.
- The earth's total allotment of water has a volume of about 344 million cubic miles. Of this:
 - 315 million cubic miles (93%) is sea water!
 - 9 million cubic miles (2.5%) is in aquifers deep below the earth's surface.
 - 7 million cubic miles (2%) is frozen in polar ice caps.
 - 53,000 cubic miles of water pass through the planet's lakes and streams.
 - 4,000 cubic miles of water is atmospheric moisture.
 - 3,400 cubic miles of water are locked within the bodies of living things.
- If all the world's water were fit into a gallon jug, the fresh water available for us to use would equal only about one tablespoon.
- It doesn't take much salt to make water "salty." If one-thousandth (or more) of the weight of water is from salt, then the water is "saline."
- Saline water can be desalinated for use as drinking water by going through a process to remove the salt from the water. The process costs so much that it isn't done on a very large scale. The cost of desalting sea water in the U.S. ranges from \$1 to \$16 per 1000 gallons.
- The overall amount of water on our planet has remained the same for two billion years.
- The United States consumes water at twice the rate of other industrialized nations.
- 1.2 Billion—Number of people worldwide who do not have access to clean water. 6.8 Billion—Gallons of water Americans flush down their toilets every day.
- Each day almost 10,000 children under the age of 5 in Third World countries die as a result of illnesses contracted by use of impure water.
- Most of the world's people must walk at least 3 hours to fetch water.

- By 2025, 52 countries—with two-thirds of the world's population—will likely have water shortages.
- The average single-family home uses 80 gallons of water *per person* each day in the winter and 120 gallons in the summer. Showering, bathing and using the toilet account for about two-thirds of the average family's water usage.
- The average person needs 2 quarts of water a day.
- During the 20th century, water use increased at double the rate of population growth; while the global population tripled, water use per capita increased by six times.
- Water use in the United States alone leaped from 330 million gallons per day in 1980 to 408 million gallons per day in 1990, despite a decade of improvements in water-saving technology.
- On a global average, most freshwater withdrawals—69%—are used for agriculture, while industry accounts for 23% and municipal use (drinking water, bathing and cleaning, and watering plants and grass) just 8%.
- Water used around the house for such things as drinking, cooking, bathing, toilet flushing, washing clothes and dishes, watering lawns and gardens, maintaining swimming pools, and washing cars

accounts for only 1% of all the water used in the U.S. each year.

- Eighty percent of the fresh water we use in the U.S. is for irrigating crops and generating thermoelectric power.
- More than 87% of the water consumed in Utah is used for agriculture and irrigation.
- Per capita water use in the western U.S. is much higher than in any other region, because of agricultural needs in this arid region. In 1985, daily per capita consumption in Idaho was 22,200 gallons versus 152 gallons in Rhode Island.
- A corn field of one acre gives off 4,000 gallons of water per day in evaporation.
- It takes about 6 gallons of water to grow a single serving of lettuce. More than 2,600 gallons is required to produce a single serving of steak.
- It takes almost 49 gallons of water to produce just one eight-ounce glass of milk. That includes water consumed by the cow and to grow the food she eats, plus water used to process the milk.
- About 6,800 gallons of water is required to grow a day's food for a family of four.
- The average American consumes 1,500 pounds of food each year; 1,000 gallons of water are required

UNITED STATES



(Source: Kammerer, J.C., *Largest Rivers in the United States*, U.S. Geological Survey Fact Sheet OFR 87-242 rev. 1990)

to grow and process each pound of that food.—1.5 million gallons of water is invested in the food eaten by just one person! This 200,000-cubic-foot-plus of water-per-person would be enough to cover a football field four feet deep.

- About 39,090 gallons of water is needed to make an automobile, tires included.
- Only 7% of the country's landscape is in a riparian zone—only 2% of which still supports riparian vegetation.
- The U.S. Fish and Wildlife Service estimate that 70% of the riparian habitat nationwide has been lost or altered.
- More than 247 million acres of United States' wetlands have been filled, dredged or channelized—an area greater than the size of California, Nevada and Oregon combined.
- Over 90% of the nearly 900,000 acres of riparian areas on Bureau of Land Management land are in degraded condition due to livestock grazing.
- Riparian areas in the West provide habitat for more species of birds than all other western vegetation combined—80% of neotropical migrant

species (mostly songbirds) depend on riparian areas for nesting or migration.

- Fully 80% of all vertebrate wildlife in the Southwest depend on riparian areas for at least half of their life.
- Of the 1200 species listed as threatened or endangered, 50% depend on rivers and streams.
- One fifth of the world's freshwater fish—2,000 of 10,000 species identified—are endangered, vulnerable, or extinct. In North America, the continent most studied, 67% of all mussels, 51% of crayfish, 40% of amphibians, 37% of fish, and 75% of freshwater mollusks are rare, imperiled, or already gone.
- At least 123 freshwater species became extinct during the 20th century. These include 79 invertebrates, 40 fishes, and 4 amphibians. (There may well have been other species that were never identified.)
- Freshwater animals are disappearing five times faster than land animals.
- In the Pacific Northwest, over 100 stocks and subspecies of salmon and trout have gone extinct and another 200 are at risk due to a host of factors, dams and the loss of riparian habitat being prime factors.

Nile (Africa): 4,132 miles

Amazon (South America): 4,087 miles

Yangtze (Asia): 3,915 miles

Huang He, aka Yellow (Asia): 3,395 miles

Parana (South America): 3,032

Congo (Africa): 2,900 miles

Amur (Asia): 2,761 miles

Lena (Asia): 2,734 miles

Mekong (Asia): 2,700 miles

Mackenzie (Canada): 2,635 miles

Niger (Africa): 2,600 miles

Yenisey (Russia): 2,543 miles

Missouri (United States): 2,540 miles

Mississippi (United States): 2,340 miles

Ob (Russia): 2,268 miles

Zambezi (Africa): 2,200 miles

Volga (Europe): 2,193 miles

Purus (Brazil): 1,995 miles

Yukon (United States/Canada): 1,980 miles

Rio Grande (United States/Mexico): 1,900 miles

St. Lawrence (United States/Canada): 1,900 mil

Sao Francisco (Brazil): 1,811 miles

Brahmaputra (India): 1,800 miles

Indus (India): 1,800 miles

Danube (Europe): 1,770 miles

(Source: Encyclopedia Britannica and others)

- A 1982 study showed that areas cleared of riparian vegetation in the Midwest had erosion rates of 15 to 60 tons per year.
- One mature tree in a riparian area can filter as much as 200 pounds of nitrates runoff per year.
- At least 9.6 million households and \$390 billion in property lie in flood prone areas in the United States. The rate of urban growth in floodplains is approximately twice that of the rest of the country.
- If all the water in the Great Lakes was spread evenly across the continental U.S., the ground would be covered with almost 10 feet of water.
- One gallon of water weighs 8.34 pounds.

RIVER LENGTHS

It's not so easy to define how long a river is. If a number of tributaries merge to form a larger river, how would you define where the river actually begins? Here, we define river length as the distance to the outflow point from the original headwaters where the name defines the complete length. (Source: Statistical Abstract of the U.S., 1986).

WORLD

Estimates for the length of the world's rivers vary wildly depending on season of the year, who is doing the measuring, the capabilities of the cartographer and his equipment and sources. However, the biggest cause of disagreeing measurements is the inclusion or exclusion of tributaries. For example, many sources lump the Mississippi and Missouri Rivers into one river system, making it one of the longest in the world. The same is true of rivers such as the Ob-Irtysh system in Asia. Considered as a whole, it is one of the ten longest rivers in the world. Removing the Irtish drops the Ob down to 15th position—assuming the rivers ahead of it also weren't measured with massive tributaries included. Here, we have tried to separate the major tributaries. You can easily find other sources that disagree with these numbers; please do not send us further questions on this.

These numbers were taken from the Encyclopedia Britannica and tributaries were separated out with help from sources like Comptons Encyclopedia and others.

SEDIMENT LOAD MEASUREMENTS

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"No matter how sophisticated, analysis and interpretation can never substitute for well collected data" (1). These words are particularly appropriate for stream sediment loads because most sediment moves during rare, large

floods that collectively account for a very small portion of time when access and safety problems are acute (1).

MODES OF SEDIMENT TRANSPORT

Two modes of sediment transport by rivers, suspended load and bed load, are usually recognized on behavioral grounds. *Suspended load* refers to the relatively fine part of the total sediment load that is transported in continuous or intermittent suspension. For sediment to remain in suspension, flow turbulence and velocity must be sufficiently great to counter the tendency of suspended sediment to settle to the riverbed. The two main factors determining sediment settling velocity are grain size and density. *Bed or traction load* refers to the relatively coarse part of the total sediment load that is transported along or close to the riverbed by rolling, sliding, or saltation. The latter term refers to sand movement in a series of hops across a riverbed in a distinct, concentrated layer.

The two modes of sediment transport are not supplementary concepts because a certain grain size can be transported in suspension by a given discharge but as bed load by another smaller discharge. Einstein et al. (2) proposed two alternative modes on the basis of experimental work on the Enoree River in South Carolina, United States. *Wash load* is the finer part of the total sediment load that is not present in bed material in significant quantities and is supplied to and transported through the measurement reach. *Bed-material load* is the coarser part of the total sediment load that is composed of grains found in bed material in appreciable quantities and is transported at a rate dependent on stream discharge. For Einstein et al.'s (2) measurement reach on the Enoree River, 96% of the measured suspended sediment during a flood was finer than 0.351 mm, and not more than 7% of the bed material was finer than 0.351 mm. The rate of transport of sediment coarser than 0.351 mm was a function of discharge, whereas the transport rate for sediment finer than 0.351 mm was related to upstream sediment supply, irrespective of whether it was transported in suspension or by traction. This limiting grain size varies among rivers depending on watershed geology, relief, rainfall, vegetation, land use, channel form, and flood peak discharge and its variability.

Vertical, Lateral and Temporal Distribution of Suspended Sediment at a River Cross Section

If suspended sediment exhibits a uniform vertical and lateral distribution and changes little during short periods of time, then the suspended load can be measured at any point in a river cross section to obtain a mean concentration. Hyperconcentrations of suspended sediment ($400 - 1000 \text{ kg/m}^3$) in the Yellow River, China, were uniformly distributed in the vertical (3). However, a large body of similar work on other rivers without hyperconcentrated suspended sediment has usually shown that the minimum concentration is found at the water surface and the maximum concentration is found near the riverbed (4,5). Furthermore, the coarsest fractions, which are usually sand, exhibit the greatest variation (5).

Nevertheless, Curtis et al. (4) reported that concentrations of particles as small as 1 or 2 μm (clay) in the Amazon River still showed slight but discernible differences between the riverbed and surface. They concluded that suspended sediment fluxes based only on surface samples are likely to contain significantly large errors.

A number of studies reviewed on the lateral variation of suspended sediment concentration at a cross section found that variations of 20% were common and that some were as large as 70% (5). This is caused by lateral variations in flow velocity and depth (and the consequent development of mobile bedforms) and by incomplete mixing of sediment-laden inflows from small tributaries into large rivers (Fig. 1). The latter effect needs to be evaluated carefully before selecting a field measurement site that should exhibit well-mixed conditions (Fig. 1).

Measured instantaneous fluctuations in depth-integrated (see below) suspended sediment concentration can vary by up to 24% (5,6). The problems of vertical, lateral, and temporal variations in suspended sediment concentration at a cross section are considerable and indicate that suspended sediment samples must be collected during a meaningful period of time from the full flow depth at multiple verticals on a river cross section.

Suspended Load. The suspended sediment load (Q_s in tonnes/day) is calculated from:

$$Q_s = 0.0864QC_s \quad (1)$$

where Q is the discharge (m^3/s) and C_s is the suspended sediment concentration (mg/L).

It is, therefore, necessary to know the discharge at the measurement site and also the mean suspended sediment concentration to calculate reliable suspended loads. For this reason, sediment load measurement sites should be located at well-mixed sections at river gauging stations.

Three requirements for an efficient suspended sediment sampler are

1. the flow velocity at the sampler intake should be the same as stream flow velocity,
2. the sampler should cause minimum flow disturbance at the sampling point, and
3. the sampler intake should be oriented into the flow in both vertical and horizontal planes (7).

Six main types of suspended sediment samplers have been identified (7–10):

1. instantaneous samplers
2. point-integrating samplers
3. depth-integrating samplers
4. single stage samplers
5. pumping samplers
6. continuous monitoring techniques.

Instantaneous samplers collect a water–sediment mixture at a single time at a single point in a vertical in the stream. Clearly, they cannot account for any temporal, vertical, or lateral variations in concentration and hence are not recommended for use in sediment load programs. However, they are appropriate for other water quality or limnological programs. Point-integrating samplers collect a water–sediment mixture at a predetermined depth in a single vertical of a river cross section during a short time period and, therefore, are time-integrated samplers. A valve is used to open the nozzle, and a separate air exhaust expels the air displaced by the sample. A series of depth settings acceptable for point-integrating samplers was found (11). The U.S. Federal Inter-Agency Sedimentation Project (6,8,10) developed a series of such samplers (USP61, USP 63, USP 72); USP 61 is illustrated in Fig. 2.

Depth-integrating samplers continuously collect a water–sediment mixture at a rate proportional to the flow velocity from the water surface to near the bed as they are lowered and raised at a uniform rate at a series of verticals at a river cross section (6,9,10). The resultant suspended sediment concentration is a time- and velocity-weighted mean. Gregory and Walling (7) recommended that the vertical transit rate (the rate of lowering and raising) should not exceed 40% of the maximum flow velocity in the vertical. The intake nozzle diameter can also be changed to ensure that the sample bottle capacity is not exceeded for deep and/or fast streams. The U.S. Federal Inter-Agency Sedimentation Project (6,8,10) developed a series of such samplers (USD 43, USDH 48, USDH 59, USDH 75, USDH76, USDH 81, USD 49, USD 74, USD 77); USDH 48 and 59 are illustrated in Fig. 2. There is an unsampled zone about 0.1m high immediately above the bed for depth-integrated samplers because of the inclined sample bottle (Fig. 2) to ensure that bed material is not scooped into the nozzle when the sampler reaches the bed. Depth-integrated samplers have been the accepted standard for hand field measurements of suspended load for the last 50 years (6,7,9,10).

Depth-integrated suspended sediment sampling is conducted according to either the equal transit rate (or equal width increment) method or the equal discharge



Figure 1. Upstream view of the Sacramento River at Knights Landing, California, showing a muddy plume originating from the tributary in the middle left of the photograph which follows the right bank for some distance before completely mixing with the less turbid mainstream flow (W.D. Erskine photograph).

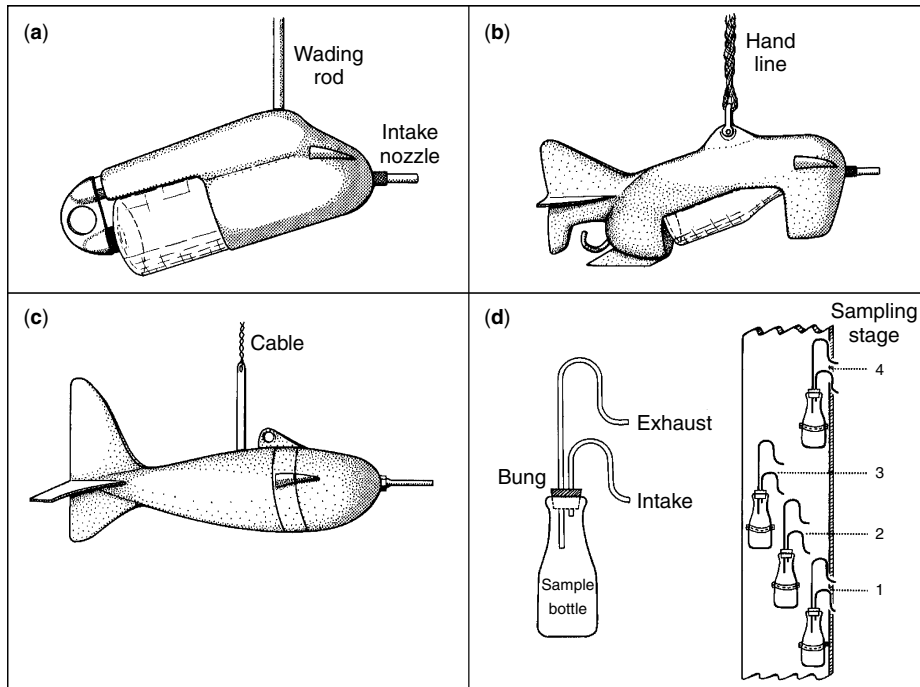


Figure 2. Suspended sediment samplers from Reference 7. (a) USDH 48 depth-integrating sampler; (b) USDH 59 depth-integrating sampler; (c) USP 61 point-integrating sampler which can also be used as a depth-integrating sampler if the valve is left open; and (d) single-stage (rising stage) samplers.

increment method (8,9). The equal transit rate method involves moving the sampler at a constant rate between the water surface and the bed at each of a series of verticals equally spaced across the channel. No prior knowledge of the velocity pattern is required at the sampling site to obtain a mean suspended sediment concentration for the whole section. Edwards and Glysson (8) developed guidelines for determining the number of verticals needed to achieve a given relative standard error based on the percentage of sand in the sample and an index of variability computed from the flow velocity and depth. The equal discharge increment method involves dividing the cross section into a series of segments of equal discharge and collecting depth-integrated samples at the centroid of each segment. Clearly, the latter method can be used only when complete velocity–area gauging has also been completed and calculated. This is not always the case.

Single-stage samplers are a type of automatic point-integrating device (10). They consist of a sample bottle equipped with intake and exhaust tubes which fills automatically when the stream first reaches the stage that submerges the sampler (Fig. 2). Such samplers usually do not collect a sample representative of the mean suspended sediment concentration at the cross section and can be contaminated by postinundation flows which can pass through the sampler.

Pump samplers remotely obtain detailed time series variations in suspended sediment concentrations at a fixed point in the stream and are required for accurate load determinations (9,10,12). Sampler intakes should be set either at a fixed position characterized by sufficient flow turbulence to ensure minimum variation in concentration throughout the cross section (13) or on a depth proportional intake boom (12). The latter consists of a hinged boom attached to the riverbed at a pivot and

suspended by a float; the intake is set at an appropriate proportion of the flow depth (12). The boom should be at least twice as long as the maximum expected stage, and the intake should be set at an acceptable depth, 0.6 was adopted by Eads and Thomas (12). The boom rises and falls with the flood stage and can be depressed by transported large woody debris to avoid damage because it is oriented downstream. Nevertheless, a single pump sampler intake still equates to a point-integrating approach to sampling (12), and the representativeness of the sample intake point must always be determined.

Pump samplers can be programmed to collect water–sediment samples at specific turbidity thresholds (14) or by various stratified or variable probability plans that reduce the variance in the sediment load. Thomas and Lewis (15) evaluated flow-stratified, time-stratified, and selection-at-list time (SALT) sampling plans and found that

- time-stratified sampling generally gives the smallest variance in storm sediment loads;
- flow-stratified and SALT sampling have difficulties only with small storms because of lack of data; and
- flow-stratified sampling produces the lowest variance for longer, more complex hydrographs that have numerous peaks.

A number of simultaneous pump and depth-integrated samples should be obtained to determine if the pump sample intake position is appropriate to produce suspended sediment concentrations that are equivalent to, or can be converted to equivalence with, depth- and width-integrated samples (12,13). Pumped-sample suspended sediment concentrations obtained from a single, fixed-position intake in a zone of turbulent flow generally varied

within 10% of depth-integrated samples without a consistent trend to over- or underestimate concentrations (13). Pump-sample suspended sediment concentrations collected by a constant depth proportion intake were closely related to depth- and width-integrated samples ($r^2 = 0.98$) but pumped concentrations were higher for concentrations above approximately 100 mg/L (12). These results highlight that pump samplers should be used for automatic suspended sediment sampling, providing that the resultant concentrations are equivalent to, or can be converted to equivalence with, depth- and width-integrated samples.

The most common continuous monitoring technique uses turbidity and measures the attenuation or scattering of an incident beam of radiation or light. Turbidity can be measured with an *in situ* probe at frequent time intervals, and the resultant values can often be converted to depth- and width-integrated suspended sediment concentrations (13,14,16–18). Reported r^2 values for relationships between depth-integrated suspended sediment concentration and turbidity can be as high as 0.90 to 0.99 (13,15). However, such relationships are usually site-specific and more reliable where suspended sediment is dominated by silt and clay (14,16).

Bed Load. A range of different portable and permanent bed load samplers has been developed for use in sand- and gravel-bed rivers, but their performance has often been poor (19). Portable samplers include baskets (7,19), trays (7,19), pressure-difference samplers (19–23), acoustic meters (19), and pumps (24). Permanent samplers are associated with major in-stream structures and include vortex tubes (25), pits (19), slot with a conveyor belt emptying into a continuous weighing hopper (26,27), and the Birkbeck bed load sampler (28). The latter consists of slots in the riverbed fitted with boxes resting on water-filled pressure pillows and combined with pressure transducers for stream height measurement (28). Most early samplers were unsatisfactory because of inaccurate measurements, variable and poor sampling efficiency (19), inconsistent sampler placement on the riverbed, and the oscillatory nature of bed load transport (19,29). The slot with a conveyor belt (26,27) is the most innovative and reliable technique devised to date but is also very expensive.

The Helley–Smith pressure-difference bed load sampler was designed and developed by the U.S. Geological Survey, Water Resources Division to be compatible with depth-integrated suspended sediment samplers (22) and is one of the most accurate (20,21,23). For sediments whose diameters are between 0.5 and 16 mm, the sampler has a near-perfect sampling efficiency that is higher for finer sediment and lower for coarser sediment (20,21). Sample bags can be filled to about 40% capacity with sediment larger than the mesh size without reducing hydraulic efficiency (20,21). Larger than standard sampling bags and increased bag mesh size improved sampling efficiencies in streams that have high sand and organic matter fluxes because of reduced bag plugging (23,30,31). Short sampling times also increase sampling efficiency in the same streams (31). Thin-walled samplers (1.5 mm) also measure larger sand fluxes than thick-walled (6.3 mm)

samplers due to local hydraulic effects (32). To operate efficiently, the sampler must have a reasonable fit between the sampler base and the riverbed and will not sample satisfactorily when the bed is highly irregular (21). The inlet orifice diameter of the original Helley–Smith sampler (76.2 mm) closely approximates the unsampled zone between the intake nozzle and the bed for a depth-integrated suspended sediment sampler (22). Therefore, depth-integrated suspended load samplers collect intermittently suspended bed material and wash load whereas pressure-difference bed load samplers collect sediment moving by rolling and sliding, bedform migration, and saltation.

There is considerable temporal and spatial variability inherent in bed load transport rates (20,21,29,32,33) which makes field measurements difficult. Bed load fluxes measured at a fixed sampling point during constant discharge on a sand-bed stream that has dune bedforms range from near zero to approximately four times the mean rate, and about 60% are less than the mean (29,32,33). Field measurements indicate that highest transport rates occur in the center of the channel and diminish near the margins (32). The U.S. Geological Survey provisional method for bed load gaugings entails two complete traverses of a cross section and at least 20 equally spaced, point measurements for 30–60 s at least 0.5 m apart on each pass (21). Although instantaneous bed load fluxes are highly variable, it is usually possible to define a mean flux for a given discharge, provided thorough field measurements are made (21,32). Marked differences in bed load fluxes and their relationship to discharge occur between pools and riffles (21).

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SEDIMENTATION

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Sedimentation is a science of interest to geologists, river morphologists, environmentalists, and engineers. Geologists are interested in the formation, mineral content, and age of sedimentary rocks. River morphologists are interested in the formation of rivers, their patterns, and profiles. Environmentalists are interested in the impacts of sediments on water quality, aquatic habitat, and ecosystems. Engineers are interested in the movement of sediment and its impact on the engineering design and operation of hydraulic structures. Due to these differences of interests and emphases, sedimentation studies can vary from qualitative to quantitative or a combination of both.

This article emphasizes the hydraulics of sediment transport. Due to the complexity of sediment transport, only basic assumptions and approaches used by engineers are summarized here. More detailed descriptions and analyses of sediment transport, scour, and deposition can be found in text books such as those by Julien (1), Yang (2), Simons and Sentürk (3), and Yalin (4).

Sediment transport can be classified as bed load, suspended load, and total load. Two basic approaches are used to study sediment transport, the probabilistic and deterministic approaches. Einstein (5) developed the most well-known probabilistic theory for sediment transport. Engineering applications of Einstein's transport functions are limited due to laborious computation procedures and extensive field data requirements. However, the modified Einstein method (6) has been used by engineers to estimate total bed-material load where direct bed load measurements are difficult to obtain.

The deterministic approach assumes that there is a one-to-one functional relationship between the dependent variable of sediment transport rate, or concentration, and independent variables. Commonly used independent variables are water discharge, average flow velocity, water surface or energy slope, shear stress, stream power, and unit stream power.

Figure 1 shows that unit stream power has the strongest one-to-one correlation with sediment discharge.

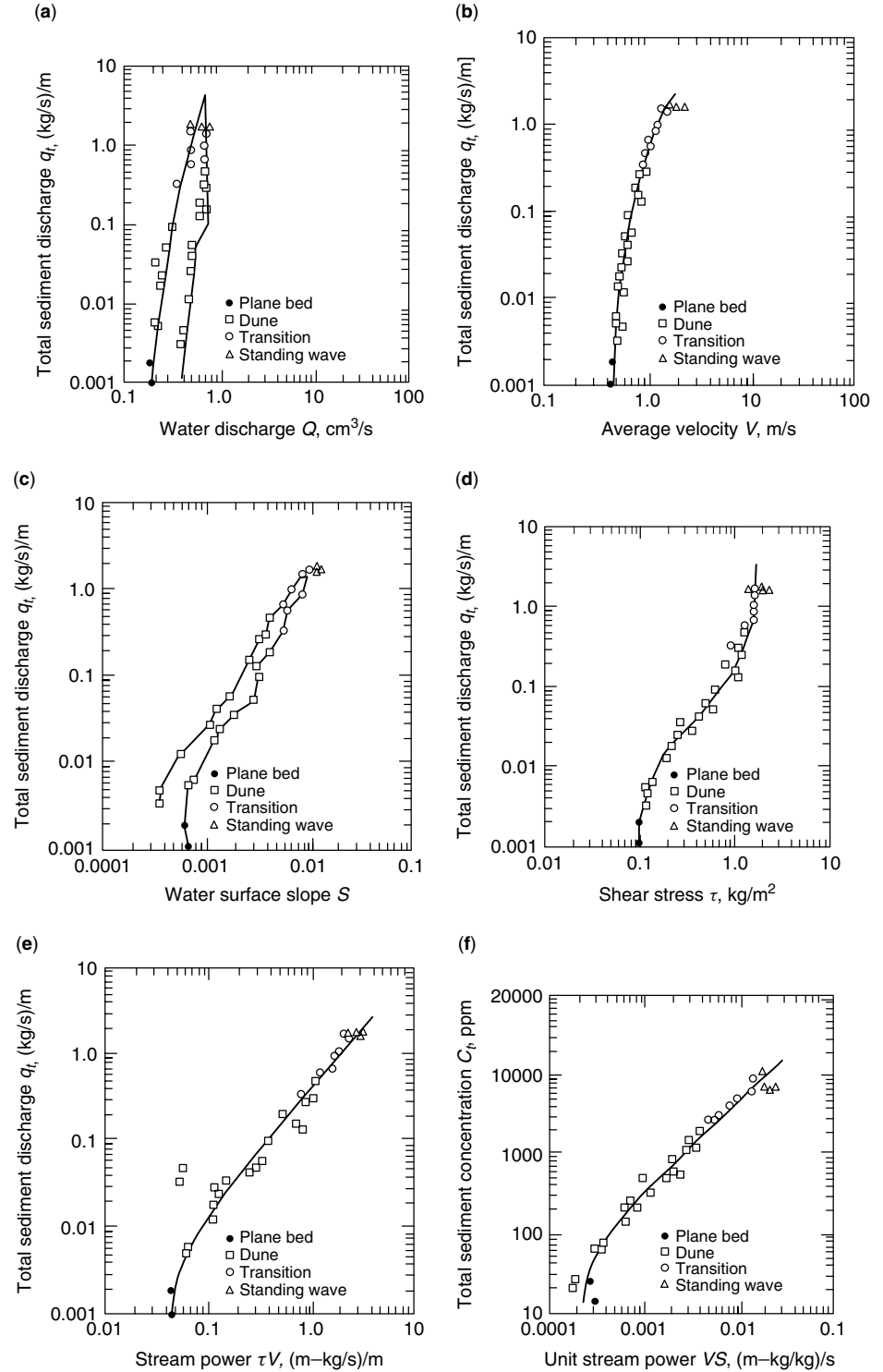


Figure 1. Relationships between total sediment discharge and (a) water discharge, (b) velocity, (c) slope, (d) shear stress, (e) stream power, and (f) unit stream power, for 0.93-mm sand in an 8-ft wide flume (7).

Yang's 1973 (8) dimensionless unit stream equation is

$$\log C_{ts} = 5.435 - 0.286 \log \frac{\omega d}{\nu} - 0.457 \log \frac{U_*}{\omega} + \left(1.799 - 0.409 \log \frac{\omega d}{\nu} - 0.314 \log \frac{U_*}{\omega} \right) \log \left(\frac{VS}{\omega} - \frac{V_{cr}S}{\omega} \right) \quad (1)$$

where C_{ts} = total sediment concentration in ppm by weight, VS = unit stream power, U_* = shear velocity, ν = kinematic viscosity of water, ω = fall velocity of sediment, d = median sediment particle diameter, V = average flow velocity, S = slope, and V_{cr} = critical flow velocity at incipient motion.

The independent variables in Eq. 1 can be measured directly in the field except the fall velocity of sediment.

The fall velocity can be obtained from figures based on experimental results recommended by the U.S. Inter-Agency Committee on Water Resources, Subcommittee on Sedimentation (9). The fall velocity can also be computed by Rubey's 1933 formula (10).

The sediment concentration or sediment load computed from different formulas can vary drastically from each other and from measurements. Figure 2 shows comparison among the computed total sediment discharges using different formulas and the measured results from the Niobrara River near Cody, Nebraska. Table 1 summarizes the ratings of some of the commonly used sediment transport formulas by the American Society of Civil Engineers Task Committee for users to consider.

Riverbed sediment particles are not uniform in size. Computations of sediment transport by size fraction are necessary to obtain realistic results for engineering purposes. Computer models are often used to simulate and predict the process of scour, transport, and deposition in rivers and reservoirs. The U.S. Army Corps of Engineers' HEC-6 (13) model and the Bureau of Reclamation's GSTARS 2.1 (14) and GSTARS3 (15) are some of the commonly used models in the public domain for solving river engineering and river morphology problems where

Table 1. Summary of Rating of Selected Sediment Transport Formulas^a

Formula Number	Reference	Type	Comments
1	Ackers and White	Total load	Rank = 3 ^b
2	Engelund and Hansen	Total load	Rank = 4
3	Laursen	Total load	Rank = 2
4	MPME ^c	Total load	Rank = 6
5	Yang	Total load	Rank = 1, best overall predictions
6	Bagnold	Bed load	Rank = 5
7	Meyer-Peter and Müller	Bed load	Rank = 7
8	Yalin	Bed load	Rank = 8

^aReference 12.

^bBased on a mean discrepancy ratio (calculated over observed transport rate) from 40 tests using field data and 165 tests using flume data.

^cMeyer-Peter and Müller's formula for bed-load and Einstein's formula for suspended load.

the movement of sediment, scour, and deposition are subjects of concern. The Bureau of Reclamation's models and user's manuals can be obtained by going to <http://www.usbr.gov/pmts/sediment/> and following the links therein.

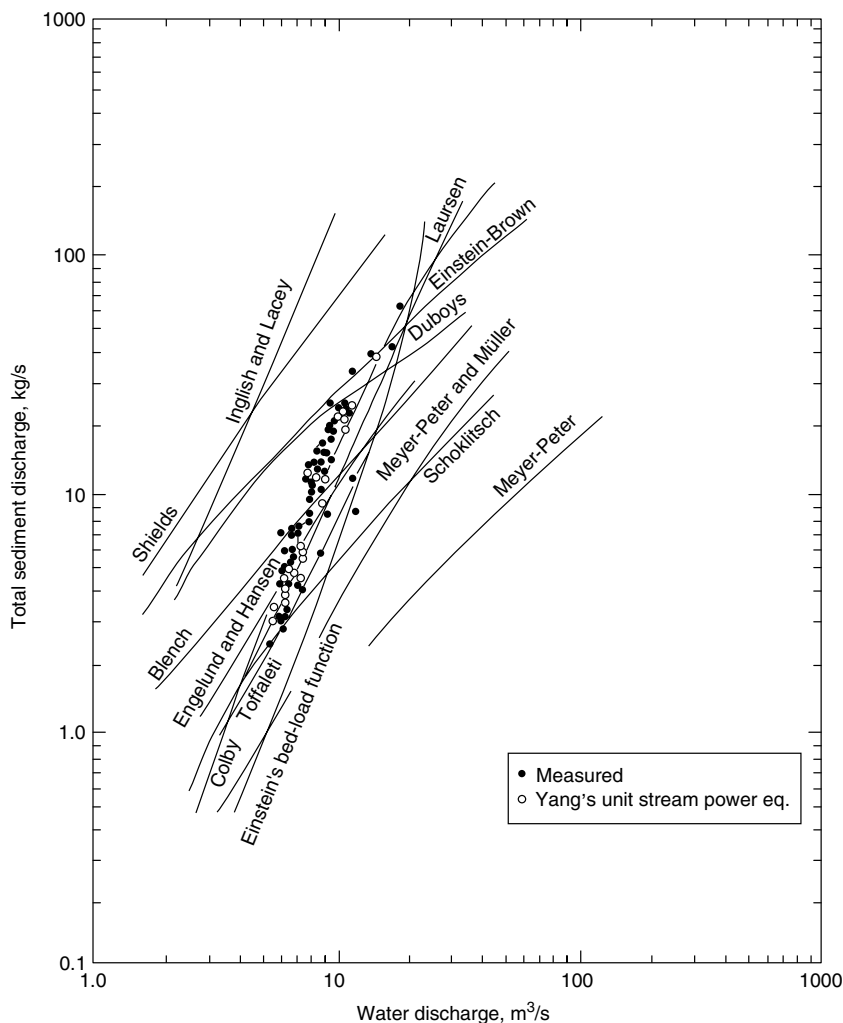


Figure 2. Comparison among measured total sediment discharge of the Niobrara River near Cody, Nebraska, and computed results of various equations (11).

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SEDIMENTATION AND FLOTATION

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SEDIMENTATION

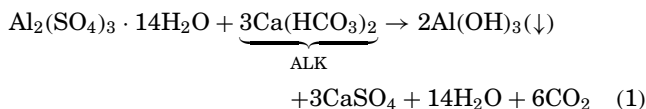
Overview

Waters and wastewaters typically contain particulate matter ranging from a few microns to centimeters in diameter, although the actual size distribution is often highly variable. One of the most common methods for

the removal of particulate matter from waters and wastewaters is simple gravity settling, often referred to as “sedimentation.”

In water and wastewater treatment, four general classes of sedimentation can occur: Type 1, discrete settling; Type 2, flocculent settling; Type 3, hindered or “zone” settling; and Type 4, compression settling. In Type 1 settling, particles settle individually with no interparticle interactions. Consequently, no change in particle size occurs (i.e., floc formation). This system is often analyzed using *Stokes' Law*, based on a force balance on a single particle settling in a viscous medium under laminar flow conditions. The development and application of this method of analysis is demonstrated in a subsequent subsection.

In flocculent settling, small particles aggregate to form larger particles. As the particles increase in size, the particle settling velocity increases (for particles of equal density). In order to make this process occur, chemicals are added to induce particle coalescence/coagulation and to promote the formation of “floc.” In water treatment, a coagulant such as [hydrated] aluminum sulfate (“alum”), ferric sulfate, ferric chloride, silica, clays, and polymers can all be used to promote the formation of larger particles. In order to determine the optimum coagulant dose for a particular water, bench-scale tests must be performed. Further, it should be noted that the effectiveness of individual coagulants can be highly pH-dependent. For example, the formation of $\text{Al}(\text{OH})_3(\downarrow)$ through the addition of alum requires $5 \leq \text{pH} \leq 8$ and sufficient alkalinity for the reaction to initiate, as presented in Eq. 1. (Note that alum can also be hydrated with $18 \text{ H}_2\text{O}$.)



Hindered or “zone” settling occurs when interparticle interactions inhibit the discrete settling of adjacent particles. In this case, particles tend to remain in a fixed position with respect to one another. Consequently, the agglomerated particles settle as a zone and a distinct solid-liquid interface develops at the top of the settling mass. This process occurs without the addition of chemical coagulants. In Type 4 or “compression settling”, a structure is formed in which settling occurs via gravity compression of the structure, because of the weight of particles. Compression settling typically occurs in samples with high solids concentration, such as the secondary clarifier in wastewater treatment or in sludge thickening facilities (1,2).

Analysis and Modeling

To develop an understanding of the physics involved in sedimentation, it is useful to examine the simplest case of discrete settling. Based on the general force balance presented in Fig. 1, the velocity of a solid spherical particle of a given diameter and density settling in a viscous medium can be determined according to Eq. 2a.

$$F_{\text{gravity}} - F_{\text{buoyant}} - F_{\text{drag}} = m_p a_s \quad (2a)$$

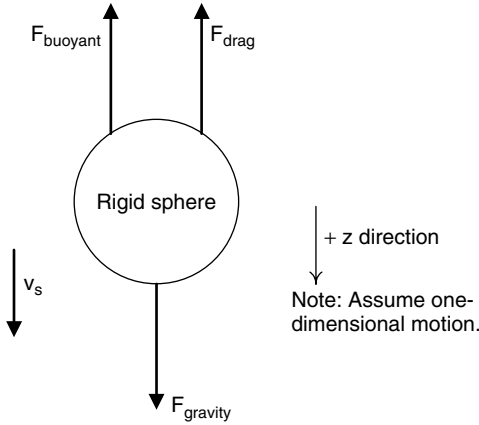


Figure 1. General force balance for a solid spherical particle settling in a viscous medium.

where F_{gravity} = gravitational force, F_{buoyant} = buoyant force, F_{drag} = drag force, m_p = particle mass, and a_s = acceleration of settling particle. If it is presumed that a terminal settling velocity has been reached (i.e., $a_s = 0$), the force balance can be rewritten as:

$$F_{\text{gravity}} - F_{\text{buoyant}} - F_{\text{drag}} = m_p a_s = m_p \underbrace{\frac{dv_s}{dt}}_0 = 0 \quad (2b)$$

The terms in the force balance can be written explicitly as presented in Eqs. 3a, 3b, and 3c.

$$F_{\text{gravity}} = V_p g \rho_p \quad (3a)$$

$$F_{\text{buoyant}} = V_p g \rho_w \quad (3b)$$

$$F_{\text{drag}} = C_D A_p \rho_w \frac{v_s^2}{2} \quad (3c)$$

where V_p = particle volume, g = gravitational constant (9.8 m/s^2), ρ_p = density of particle (typical value = $1,400 \text{ kg/m}^3$), ρ_w = density of water ($1,000 \text{ kg/m}^3$ under standard conditions), C_D = drag coefficient, A_p = cross-sectional area of the particle, and v_s = particle settling velocity.

Based on the general force balance presented in Fig. 1, and using Eqs. 2 and 3, the settling velocity for spherical particles of a given diameter and density can be determined as:

$$v_s = \sqrt{\frac{4}{3} \frac{g(\rho_p - \rho_w)d_p}{C_D \rho_w}}, \text{ Re} \geq 1 \quad (4)$$

As only one-dimensional motion is considered, vector notation has been dropped and the sign convention presented in Fig. 1 is used throughout.

The drag coefficient for rigid spheres can be calculated according to Eq. 5, where C_D is a function of the hydraulic flow regime, established using the Reynolds number, Re , given in Eq. 6.

$$C_D = \frac{24}{\text{Re}} + \frac{3}{\sqrt{\text{Re}}} + 0.34 \quad (5)$$

$$\text{Re} = \frac{v_s d_p \rho_w}{\mu_w} = \frac{v_s d_p}{\nu_w} \quad (6)$$

When spherical particles settle in a viscous medium under laminar flow conditions ($\text{Re} < 1$), the first term in Eq. 5 predominates (i.e., $C_D \simeq 24/\text{Re}$) and the expression for settling velocity becomes more simple:

$$v_{si}(\text{Stokes}') = \frac{g(\rho_p - \rho_w)d_{pi}^2}{18\mu_w}, \text{ Re} < 1 \quad (7)$$

This form of the settling velocity is known as “Stokes’ Equation.” As most waters and wastewaters contain particles of many different sizes, an additional subscript “i” is typically added to denote particles of the same size in an otherwise heterogeneous distribution. Consequently, a different settling velocity would have to be calculated for different size particles (1,2).

Application

In order to adequately analyze particle removal through sedimentation, it is necessary to establish a framework for analysis. In this case, consider the longitudinal cross section of a settling basin shown in Fig. 2. Assume a uniform distribution of particles at the entrance, along which particles can enter at any height. Further, any particle that settles into the sludge zone is permanently removed from the system. Now, define the “critical settling velocity” (v_{sc}) as the settling velocity of the smallest particle that will be 100% removed. To further ease the analysis of this system, the critical settling velocity vector will be broken into its horizontal (v_h) and vertical (v_{sc}) components. Note that in sedimentation, we are primarily concerned with motion in the vertical direction, so the vertical component of the settling velocity is often labeled “ v_{sc} .” The trajectory of particles with a velocity, v_{si} , equal to v_{sc} is also presented in Fig. 2. In this case, 100% of the particles are removed from the system, regardless of their entry position at the inlet.

Similarly, the trajectory of particles with a settling velocity greater than v_{sc} is presented in Fig. 3. [The trajectory of particles with a critical settling velocity (dashed lines) has been added to Fig. 3 for reference.] In this case, 100% of the particles are removed from the system, regardless of their entry position at the inlet.

The trajectory of particles with $v_{si} < v_{sc}$ is presented in Fig. 4. By examining the trajectories of these particles,

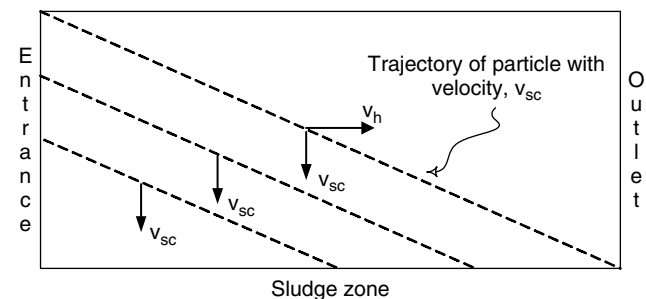


Figure 2. Trajectory of particles with a settling velocity equal to v_{sc} .

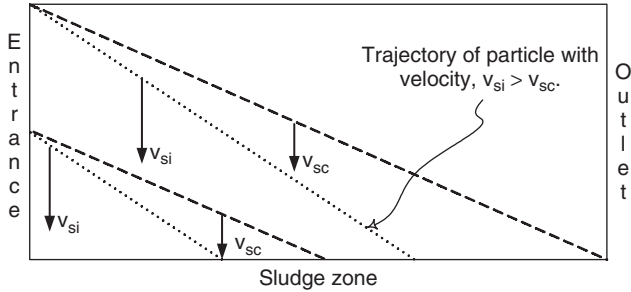


Figure 3. The trajectory of particles with a settling velocity greater than v_{sc} .

we can see that it is possible to have some particles “escape” through the outlet. Likewise, some particles may become trapped in the sludge zone, depending on their respective position at the inlet and the magnitude of v_{si} relative to v_{sc} . Consequently, when the settling velocity of an individual particle is less than the critical settling velocity, fractional particle removal will occur; the fraction of particles removed when $v_{si} < v_{sc}$ is given by Eq. 8 (1,2).

$$F_r = \frac{v_{si}}{v_{sc}} \quad (8)$$

Process Design and Implementation

In a typical water treatment process, sedimentation follows coagulation and flocculation, as presented schematically in Fig. 5. In activated sludge wastewater treatment, two sedimentation basins are used, as presented in Fig. 6. With regard to sedimentation basin design, it is common to determine the hydraulic detention time, θ_H , and hydraulic surface loading rate, S , presented in Eqs. 9 and 10, respectively. Typical design parameters for sedimentation tanks used in primary and secondary wastewater treatment are presented in Tables 1 and 2, respectively (1,3).

$$V_{\text{basin}} = \theta_H \cdot Q \quad (9)$$

$$S = \frac{Q}{A} \quad (10)$$

where V_{basin} = volume of sedimentation basin, θ_H = hydraulic detention time, Q = water/wastewater flow rate, and A = basin surface area.

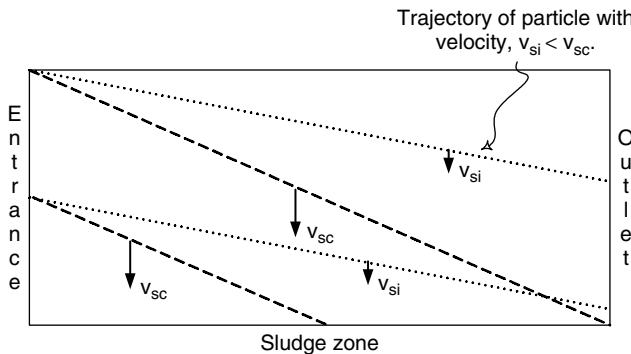


Figure 4. The trajectory of particles with a settling velocity less than v_{sc} .

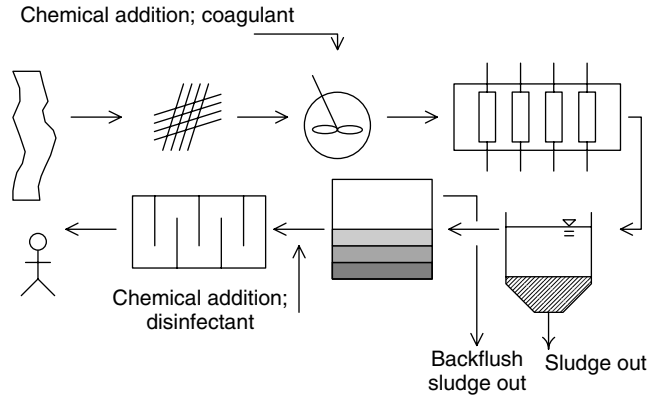


Figure 5. Schematic showing location of sedimentation in typical water treatment process.

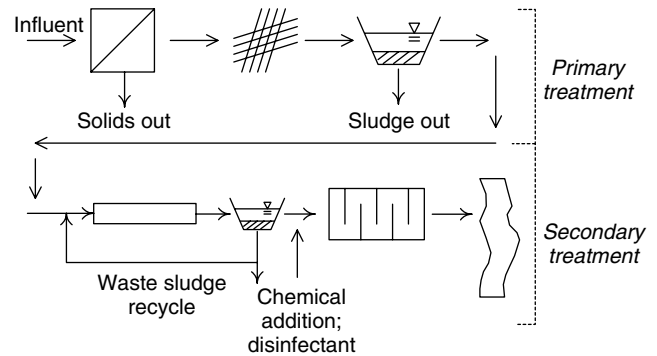


Figure 6. Schematic showing location of two sedimentation basins used in activated sludge wastewater treatment.

Table 1. Typical Design Parameters for Sedimentation Tanks Used in Primary Wastewater Treatment (1)

Primary Treatment Followed by Secondary Treatment	Range	Typical
Hydraulic detention time, hr	1.5–2.5	2.0
Average overflow rate (average flow), gal/ft ² ·d	800–1,200	
Overflow rate (peak flow), gal/ft ² ·d	2,000–3,000	2,500
Primary Treatment Followed by Waste Activated Sludge		
Hydraulic detention time, hr	1.5–2.5	2.0
Average overflow rate (average flow), gal/ft ² ·d	600–800	
Overflow rate (peak flow), gal/ft ² ·d	1,200–1,700	1,500

The data presented in Table 1 are for general reference purposes. In actual system design, factors such as water flow rate, basin depth, and desired solids removal must be considered (4).

FLOTATION

Flotation is a process for the separation of particulate matter from liquids in which bubbles are introduced into

Table 2. Typical Design Parameters for Sedimentation Tanks Used in Secondary Wastewater Treatment (3)

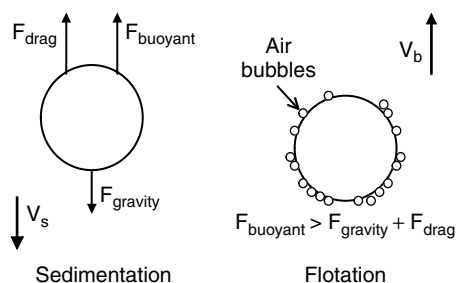
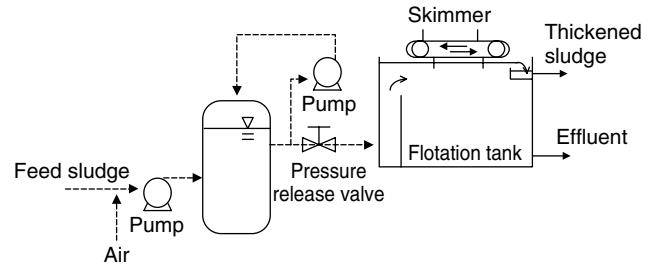
Secondary Treatment	Range
Average overflow rate (average flow), gal/ft ² · d	400–800
Overflow rate (peak flow), gal/ft ² · d	1,000–1,200
Average solids loading rate, lb/ft ² · d	0.8–1.2
Peak solids loading rate, lb/ft ² · d	2.0
Depth, ft	12–20

a wastewater and the solid particles rise with the bubbles. The particulate matter is then skimmed off the water surface and clarified water is removed from the bottom of the basin (1). A comparative schematic of sedimentation and flotation processes is presented in Fig. 7. Flotation is typically applied in cases where particles are difficult to remove via sedimentation; examples include synthetic fibers in the textile industry, wood fibers in the pulp and paper industry, and oily wastewaters (5). Flotation is also used in activated sludge thickening (1).

In general, air flotation and dissolved air flotation are the two types of flotation processes used most widely in industry. In air flotation, air is introduced via fine bubble diffusers. In this process, problems with diffuser clogging are common. Additionally, a large fraction of the operating cost is associated with the need to supply large amounts of power to force air bubbles through the diffusers.

Dissolved air flotation (DAF) is different from air flotation mainly in the conditions under which air enters the system. A schematic of a typical DAF process used to thicken activated sludge is presented in Fig. 8 (1). In dissolved air flotation, air is injected into wastewater under pressure. In typical wastewater applications, air is pressurized to 345–483 kPa (3.4–4.8 atm; 50–70 psi) (5), which results in an increase in the solubility of air in water. When the pressure is released to the atmosphere, the excess dissolved air comes out of the aqueous phase and rises. In order to enhance the particle removal efficiency of DAF processes, it is common to add chemicals that promote flocculation to the influent prior to pressurization. Reagents used for this purpose are similar to those used in coagulation/flocculation processes (e.g., alum, polymers, etc.) (1,5).

Vacuum flotation is a third technique in which wastewater is saturated with air under atmospheric pressure. A vacuum is applied to the wastewater in an enclosed tank, where the air is pulled from solution

**Figure 7.** Comparative schematic of sedimentation and flotation.**Figure 8.** Schematic of a typical dissolved air flotation process used to thicken activated sludge. (Note: no recycle).

and particles rise to the top. The application of vacuum flotation is generally limited to small operations, because higher capital and operating costs associated with maintaining a closed reactor system.

FLOTATION VERSUS SEDIMENTATION?

The selection of either sedimentation or flotation as a solid/liquid separation process depends on a large number of factors, including particle density, overflow rate, aesthetic requirements, sludge volume production rate, ease of operation, and availability of reliable design data. However, the guidelines presented in Table 3 can be used to compare and screen these processes for potential application.

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Table 3. General Guidelines/Comparison for Selection of Sedimentation and Flotation Processes

Flotation	Sedimentation
Generally favored when particles with low density (low specific gravity) are to be removed.	Substantially less complicated to operate (e.g., fewer maintenance and operating problems, requires less skilled staff than flotation processes).
Can be used when space and/or capital are limited because higher overflow rates can be applied, which results in smaller tank requirements.	Requires less power for operation.
Odor problems can be minimized because of the lack of septic conditions.	Can easily be designed from the wealth of reliable performance data available.
A thicker sludge and, consequently, a lower sludge volume is produced.	

4. Reynolds, T. and Richards, P. (1995). *Unit Operations and Processes in Environmental Engineering*, 2nd Edn. PSW Publishing, Boston, MA.
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RESERVOIR SEDIMENTATION

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The detachment and transportation of soil is termed soil erosion. Every stream carries some sediment in suspension and moves larger particles along its bed. The deposition of sediment in channels or reservoirs creates a variety of problems, such as raising stream beds, meandering and overflow along the banks, and, of course, depletion of storage capacity in reservoirs.

The sediment content in river waters varies with flow conditions. It is small in lean flow months, but it attains a maximum value during floods. The total quantity of sediment transported annually to the sea by rivers of the world is about 2×10^{10} tons or about 13.5 km^3 in volume. Worldwide, reservoirs are annually losing about 1% of their storage capacity or about $65 \text{ km}^3/\text{year}$. The rate of loss of reservoir storage in the United States is about 0.22%/year which is equivalent to 2020 million m^3/year . Based on weighted average data from 144 reservoirs in India, the annual loss of gross storage was estimated at 0.44%.

Sediment source and reservoir locations are not uniformly distributed; the problem is really severe in some countries. China has more than 80,000 reservoirs that are annually losing, on average, 2.3% of their storage capacity due to sedimentation.

INTRODUCTION

Soil erosion is considerably high in arid climates. In India, about 5333 million tonnes (16.35 t/ha) of soil is detached annually due to agriculture and associated activities. Of this, about 29% is carried away to oceans by rivers. Nearly 10% of it is deposited in reservoirs resulting in loss of 1–2% of storage capacity (1). The sediment transport of selected rivers is given in Table 1.

Each sediment particle transported by flow is affected by two dynamic forces: a horizontal component acting in the direction of flow and a vertical component due to gravity; there is also the force of water turbulence. The specific gravity of soil materials is about 2.65, so particles of suspended sediment tend to settle to the channel bottom, but upward currents in the turbulent flow counteract gravitational settling. The sediment inflow and outflow in natural river reaches are mostly in balance. A reservoir changes the flow characteristics and its sediment transport capacity. Because the reservoir width is much bigger than the river width, the velocity of flow entering into it decreases tremendously and there is a dampening of turbulence. Consequently, the flow is unable to transport

Table 1. Sediment Transported by Selected Rivers^a

River	Drainage Area, km^2	Average Sediment Concentration, kg/m^3	Erosion Modulus, $\text{ton/km}^2/\text{yr}$
Nile	2,978,000	1.25	37
Missouri	1,370,000	3.54	159
Colorado	637,000	27.5	212
Indus	969,000	2.49	449
Irrawaddy	430,000	0.70	695
Brahmaputra	666,000	1.89	1090
Red	119,000	1.06	1092
Ganges	955,000	3.92	1519
Liaohe	166,300	6.86	240
Yangtze	1,807,200	0.54	280
Yellow	752,400	37.6	2,480

^aAdapted from Reference 2.

all the sediments, and the particles begin to deposit. First, the larger suspended particles and most of the bed load are deposited at the mouth of the reservoir. The smaller particles remain in suspension for a long time and some may leave the reservoir with water.

RESERVOIR SEDIMENTATION

The accumulation of sediments is one of the principal factors that threaten the longevity of reservoirs. Sometimes a project is not constructed just because the silting rate is so high that the reservoir will fill up before the investment is fully recovered.

The ultimate destiny of all reservoirs is to be filled by sediment. There are instances of reservoirs filling up within a few years of their operation. The Sanmexia dam was the first major dam on the middle reaches of the Yellow River. In the first 18 months after dam closure, 1.8 billion metric tons of sediment accumulated in the reservoir, representing a trap efficiency of 93% (3). The Xinghe Reservoir in Shaanxi Province took 2 years to construct but only 1 year to fill with sediment.

The right approach to solving a reservoir sedimentation problem has three segments: (1) collect and analyze field data, (2) set up appropriate models, and (3) develop an operational policy for the reservoir. Deposition and scouring may differ considerably when different operating policies are adopted.

The sediment deposits in a reservoir can be divided in three groups: topset beds, foreset beds, and bottomset beds (see Fig. 1). Topset beds are composed of large size sediment deposits but may also have fine particles. These extend up to the point where the backwater curve ends. The downstream limit of the topset bed corresponds to the downstream limit of bed material transport in the reservoir. These deposits cause a minor reduction in reservoir storage capacity. The foreset deposit is the face of the delta deposit advancing toward the dam. It is a transition zone that has steeper slopes and decreasing grain size. The bottomset beds consist of fine sediments that are deposited beyond the delta by turbidity currents or nonstratified flow. This pattern of deposits may change due to reservoir drawdown, slope failures, and extreme floods.

For proper reservoir management, knowledge about the sediment deposition pattern in various zones is

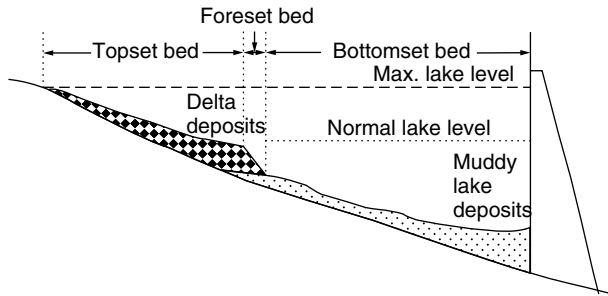


Figure 1. Sediment deposit zones in a reservoir.

essential. Timely remedial measures can be undertaken with correct knowledge of the sedimentation processes in a reservoir. A common procedure for tackling a sediment problem is to designate a portion of reservoir storage for sediment accumulation. But this approach does not solve the problem; it merely postpones the date when it becomes serious.

Factors Influencing Reservoir Sedimentation

Two dominant factors that influence the rate of silting in a reservoir are (1) capacity to inflow ratio (CIR) and (2) sediment content in the water flowing in. The other factors are the texture and size of the sediment, the trap efficiency of the reservoir, and the method of reservoir operation. The CIR is the ratio of reservoir storage capacity to mean annual inflow. A reservoir whose CIR is more than 50% is considered hydrologically large, may have significant carryover, and its trap efficiency will also be large. Note that the sediment inflow depends on the catchment area, too. All other things remaining the same, a dam of the same capacity in an upper catchment will have a higher rate of silting compared to a dam of the same design but constructed lower down in the valley.

The two principal factors mentioned above have a complete range of interplay. A reservoir that has a small CIR and small sediment inflow and one that has a large CIR and large sediment inflow may have more or less the same rate of sedimentation. A high rate of silting can also be expected from a high CIR and a high sediment content in the inflow. On the other hand, a high CIR and low sediment content in inflows will result in a low rate of silting.

Trap Efficiency

The trap efficiency of a reservoir is the ratio of sediment retained in the reservoir to the sediment inflow. It depends primarily on the sediment characteristics, the detention time of the inflow, the method of reservoir operation, and the age of the reservoir. A small reservoir on a large stream passes most of its inflow so quickly that the finer sediments are discharged downstream. A larger reservoir, on the other hand, may retain water for several years and the outflow from it may be completely devoid of suspended sediments. The trap efficiency of a reservoir decreases with age as the reservoir capacity is reduced by sediment accumulation. Thus complete filling of the reservoir may require a very long time.

The trap efficiency can be computed from the sediment inflow and outflow data. Brune (4) analyzed data from 44 reservoirs in the States; 40 were normal ponded reservoirs whose catchment areas vary from 0.098 sq. km to 478,110 sq. km and CI ratios range from 0.0016 to 2.05. The database also included two desilting basins and two semidry reservoirs. This analysis revealed that the laws of sediment deposition are the same for all types of reservoirs and the factors influencing the trap efficiency are independent of the size of the reservoir. Brune (4) presented a set of envelope curves between CIR and trap efficiency (see Fig. 2) that shows a semilogarithmic curvilinear relation between trap efficiency and CIR.

Sedimentation and Life of a Reservoir

The term *life of a reservoir* appears to be a misnomer because reservoirs do not have a single well-defined life which denotes two functional states: *ON* and *OFF*. Rather, they show a gradual degradation of performance. The 'life' of a reservoir for different purposes will be affected at different times. Murthy (5) defined the following terms connected to the life of a reservoir.

Useful Life. The period during which the capacity occupied by sediment does not prevent the reservoir from serving its intended primary purpose.

Economic Life. This is determined by the time after which the effect of various factors, such as physical deterioration by sedimentation and changing requirements for project services, cause the operating costs of the reservoir to exceed the additional benefits from its continuation.

Design Life. The period that is adopted for economic analysis. This is either the useful life or the shorter of the expected economic life or fixed span of life (say 50/100 years).

Full Life. The number of years required for full depletion of reservoir capacity by sedimentation.

LOSS OF STORAGE CAPACITY

Observations show that in reservoirs that have small sluicing capacity with respect to normal floods and have no upstream reservoirs, the siltation rate is comparatively high in the first 15–20 years and thereafter it falls off and may ultimately become negligible. From the data

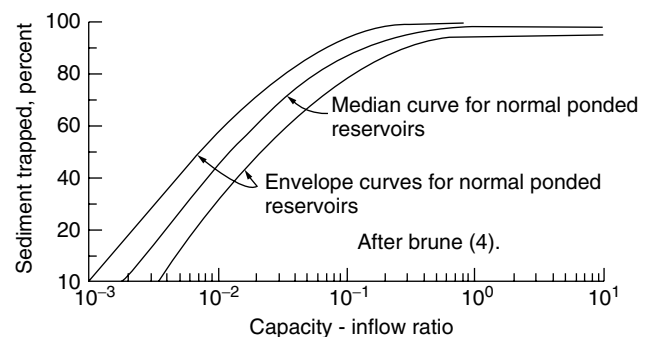


Figure 2. Brune's curves (4).

of reservoir capacity surveys, Shangle (6) found that the sedimentation rates in major reservoirs (storage > 100 million m³) in India that have completed more than 50 years of their useful lives varied from 0.30 to 4.89 Ha-m/100 sq. km/year. The rate for those major reservoirs that have completed less than 50 years of their useful lives varied from 0.34 to 27.85 Ha-m/100 sq. km/year. Deyi and Fan (7) derived an empirical formula to determine the average annual loss rate of reservoir capacity.

The rate of siltation of a reservoir normally shows a falling trend with time. A plausible explanation is that the obstruction by the dam causes the dips and flanks of the storage basin to fill up with silt in early years. A stage comes when the river section adjusts itself to carry the normal discharge and disposal of the suspended load in the area of the reservoir is harmonized with the condition of the flow. Besides, the progressive development of deltas above the reservoir helps in trapping some of the silt load. Shrinkage and settlement of deposited silt also takes place with time due to superimposed loads of additional silt. This results in reduction in silt volume. However, a complete explanation of this behavior is not available.

Unit Weight of Sediments Deposited in a Reservoir

The unitweight of sediments is the dry weight (kg) per unit volume (cubic meter) of the material. It has been observed that the density of deposited sediment varies from 500–2000 kg/m³. The important factors that influence the unit weight of deposited sediment are the manner of reservoir operation, the texture and size of sediment particles, and the consolidation rate. The reservoir operation is the most influential of these factors. If the water level of a reservoir is lowered from time to time, the deposited sediment is exposed to the Sun and air and gets dense. There may be considerable time for consolidation before floods. The degree of consolidation depends on the weight of overlying material, its exposure, sediment size, and time. The reservoir is always filled that has a low density of deposit. Power and irrigation reservoirs belong to the intermediate class. Generally, lower densities are observed in the vicinity of a dam under submerged conditions whereas higher densities are noticed in the upstream portions of reservoirs.

Lara and Pemberton of U.S.B.R. developed a method for estimating the initial unitweight of sediment deposits when the size analysis of the incoming sediment and the proposed reservoir operating schemes are known. Reservoir operations were classified in different types as follows:

Reservoir Type	Reservoir Operation	W_c	W_m	W_s
I	Sediment always submerged or nearly submerged	26	70	97
II	Normally moderate to considerable reservoir drawdown	35	71	97
III	Reservoir normally empty	40	73	97
IV	Riverbed sediments	60	73	97

The unitweight of the sediment deposits can be estimated from:

$$\gamma = 16.05(W_c P_c + W_m P_m + W_s P_s)$$

where γ is unitweight in kg/m³; P_c, P_m, P_s are percentages of clay, silt, and sand, respectively, of the incoming sediment; and W_c, W_m, W_s are the coefficients of clay, silt, and sand, respectively, which may be obtained from the table above (15).

Distribution of Sediments in Reservoirs

Sediment distribution in a reservoir is important, and this requires careful consideration in planning and design stages. The pattern of sedimentation helps in predicting the extent to which services will be affected at various times and the remedial actions to be taken. The designer is interested to know the height of sediment accumulation to fix the sill elevation of the outlets, the penstock gate elevation, and to estimate the region where a delta would be formed and the consequent increase in backwater levels. Finally, the pattern is necessary to plan for recreational facilities. Four types of distribution patterns of deposits have been identified (3):

Delta Deposits. These deposits contain the coarsest fraction of the sediment load that is rapidly deposited in the zone of inflow.

Wedge-shaped Deposits. These are thickest at the dam and become thinner moving upstream.

Tapering Deposit. These occur when deposits become progressively thinner moving toward the dam.

Uniform Deposits. The thickness of these deposits is more or less uniform in the reservoir. Such deposits are not very common.

The consequences of reservoir sedimentation are gradual reduction in benefits. The extent of loss depends on the type and nature of the purposes being served and the rate of loss of storage capacity. The loss turns out to be really high when the replacement cost of the lost storage is considered.

A commonly used empirical method to estimate the new reservoir profile is discussed next.

Empirical Area Reduction Method

Based on the analysis of the sediment survey data of many reservoirs in the United States, it was found that the sediment distribution pattern depends on reservoir geometry, operation, and sediment characteristics. Depending upon the elevation–capacity characteristics, reservoirs are classified into four types: (a) gorge, (b) hill, (c) flood plain–foot hill, and (d) lake. Empirically derived sediment distribution curves are used to distribute the sediment throughout the reservoir section. The *Empirical Area Reduction* method, as revised by Lara (8), is the most popular for predicting a new reservoir bed profile. It is necessary first to estimate the amount of sediment deposited in the reservoir. The computational steps have been given in many books, such as Morris and Fan (3).

RESERVOIR SURVEYS

Sediments accumulated in an existing reservoir can be determined by periodically running sediment surveys. It is a direct measurement procedure for assessing the volume and pattern of deposits. Sediment data collected during the surveys are analyzed to determine the specific weights of the deposits, their grain size distribution, sediment accumulation rates, and reservoir efficiencies. Recent advances in technology have considerably reduced the effort to carry out reservoir surveys and analyze data.

The advantages of reservoir surveys are as follows:

1. The reservoir survey can be less costly than continuous sediment measurement at several locations in the catchment.
2. The accuracy of these surveys is usually very high, particularly if advanced equipment is used.
3. It is possible to estimate the total sediment (bed and suspended) load carried by a river.

There are some limitations of the reservoir sedimentation survey:

1. Such surveys do not provide any information about the variation of sediment yield with time and give only the total sediment accumulated since the last survey.
2. This method does not provide subcatchmentwise sediment yield.
3. This approach is not very effective where sedimentation is small as the errors of measurement may mask the true sedimentation rates.
4. To find the total sediment inflow, information about sediment outflow is also needed.

It is essential to have an accurate map of the reservoir on an appropriate scale before commencing a survey. Important reservoir features, such as the full reservoir level (FRL) along the periphery, the position of the dam, outlets, inflowing streams, etc., along with other details such as the position of nearby bridges, roads, and villages should be marked on the map. Horizontal and vertical control points are fixed at a suitable interval on the reservoir circumference. Next, cross-sections are planned at a suitable spacing, depending on the reservoir size.

The contour and range methods are two basic techniques for surveying reservoirs. The selection of a method depends on the quantity and distribution of sediment indicated by field inspections, the shape of the reservoir, the purpose of the survey, and the degree of accuracy desired. The volume of accumulated sediment is computed by subtracting the revised capacity from the original capacity at a given reservoir elevation (usually the FRL).

The frequency of surveying reservoirs depends on the sediment accumulation rate and the cost of a survey. Reservoirs that have high accumulation rates are surveyed more often than those with lower rates. Generally, the reservoirs are surveyed every 3 to 10 years. Special circumstances may necessitate a change in the established schedule. A reservoir might be surveyed after a major

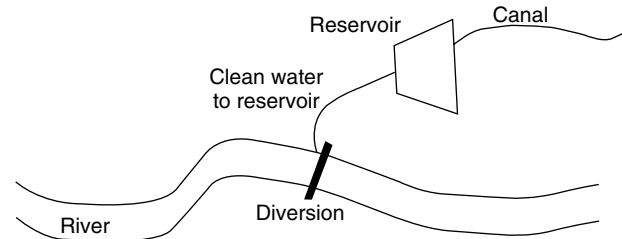


Figure 3. An offstream reservoir.

flood that has brought in a heavy sediment load. A survey may also be run following the closure of a major dam constructed upstream in the same catchment. An upstream dam reduces the free drainage area and hence reduces the sediment inflow.

Remote sensing techniques, enabling acquisition and analysis of synoptic data over a broad spectral range, are an alternative to the conventional way of data acquisition and processing. This technique is highly cost- and time-effective compared to conventional methods. The methodology for assessing reservoir sedimentation is described by Goel et al. (9).

CONTROL OF SEDIMENT INFLOW INTO A RESERVOIR

The problem of reservoir sedimentation is complex but not unmanageable. It can be largely controlled by judicious design, construction, and management. The Yellow River in China is notorious for very high volumes of sediments in its water. The Sanmenxia Dam was the first dam built in the middle reach of the Yellow River in China. After the impounding commenced, about 1.8 billion metric tonnes of sediments accumulated during the first 18 months, representing a trap efficiency of 93%. The balance between sediment inflow and outflow was restored by providing high capacity bottom outlets, and reservoir operation was substantially changed.

There are various ways to manage the sedimentation problem, and the effectiveness of an approach depends on the site conditions. A single technique will not work everywhere. Broadly, the methods are

- Control the sediment inflow into the reservoir.
- Do not allow the entering sediment to settle in the reservoir.
- Remove the settled sediment from the reservoir.

Regarding the first approach, broadly, there are three ways to prevent sediment from entering a reservoir. These are (1) constructing reservoirs away from streams, (2) physical barriers in the way of sediment movement, and (3) better watershed management to check soil erosion.

Offstream Reservoirs

If the site conditions permit, a reservoir can be constructed away from the main stream (Fig. 3). This reservoir is filled with water of low sediment concentration that is diverted from the main river. The flows that contain high amounts of sediment load are excluded from diversion. An additional advantage of an offstream reservoir is that only a desired amount of flow is diverted to it, and therefore, its capacity can be limited.

Check Dams

A check dam is a small dam a few meters high that is constructed across a stream in the headwater region to control channel erosion. Most of the incoming sediment is deposited behind the check dam and relatively clear water comes out downstream. These dams can be highly effective in controlling reservoir sedimentation. The cost of a small check dam is not very high. These dams are more efficient for sediment trapping if they are spaced farther apart. These dams are not a long-term solution to the problem because they do not control the sediment erosion, and once the reservoir behind the check dam is filled with sediments, it no longer serves the purpose.

SEDIMENT ROUTING

This is an effective method for controlling reservoir sedimentation. It includes methods to manage the hydraulic behavior of the reservoir to allow the maximum sediments to pass through. The concepts of sediment routing were developed in China. The guiding principle is *discharge the muddy water, impound the clear water*. It is an environmental friendly approach because the characteristics of sediment transport are not significantly changed.

Reservoir Drawdown

Typically, the rising limb of the hydrograph carries larger amount of sediments than the falling limb. In this approach, the reservoir water level is bought down in the beginning of a flood to pass turbid flow without deposition.

Density Currents

A density current is the gravity flow of fluid under, over, or through another fluid of approximately equal density or density whose differs by a small amount from that of the primary current. Density currents are generated when sediment-laden water enters a relatively still waterbody. The density current venting approach (Fig. 4) provides a clear unhindered path to sediment-laden flows that are released through low-level outlets.

Sediment Bypass

This arrangement, shown in Fig. 5, prohibits sediment-laden flow from entering the reservoir. It consists of a diversion structure upstream of the dam by which the flows with heavy sediment concentration are diverted to a bypass channel or conduit which joins the main river downstream of the dam. As a result, relatively 'clean' flows enter the reservoir.

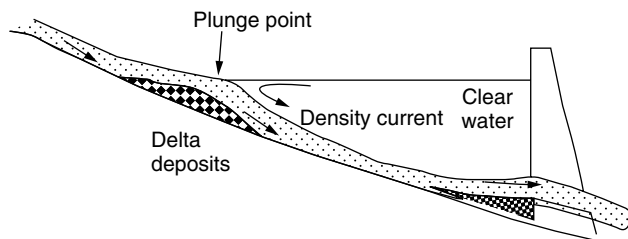


Figure 4. Venting of density current through bottom outlet in a reservoir.

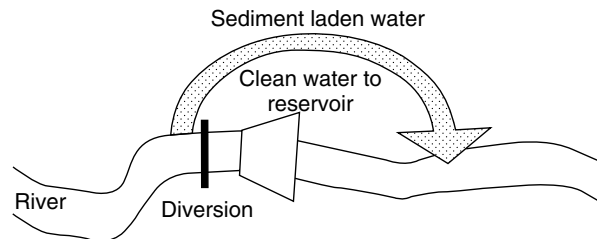


Figure 5. Sediment bypass.

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WATER FROM SATURATED RIVER SEDIMENT—SAND ABSTRACTION

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Water is available in the saturated sediment of sand-filled rivers. The removal of water from this sediment is known as sand abstraction. Despite the terminology, it is not the removal of sand but of water from a surface dry, seasonal river where the water is retained in the sediment, not abstraction of water from a flowing river.

Wherever there is erosion of land within the catchment area of a river, soil, sand, or gravel is carried into the river channel. The river becomes silted with sediment retained in depressions in the riverbed, behind rocks, and where

the slope of the river is sufficiently low for sediment to be deposited rather than washed away.

In arid and semiarid areas, the sediment that accrues is often coarse grained, and, as the river basin drains, water is retained in the sediment in useable quantities. In spite of the intermittent flow of seasonal rivers, as the rivers drain and dry out, water is held in the sediment.

BACKGROUND INFORMATION

History and Traditional Use

There have been sand-filled rivers for centuries in arid and semiarid areas where intermittent rainfall and long dry periods exist (Fig. 1). These rivers have become increasingly filled with sediment since the land has become populated and built up. As surface water dropped into the sediment, people had to dig down to find their daily water requirements. This practice continues today in many arid areas.

Traditionally, people lived in river valleys, and as the seasonal rivers dried out, they excavated inverted cone-shaped holes to expose the water level in the sediment. As the loose, unstable sand continually filled the hole, people were forced to re-excavate their holes. A natural development was to insert an opened ended drum into the river sediment as a liner. This made a 'sand well' that was more stable and did not fill with sand after every use (Fig. 2). The water abstracted was then used for domestic, livestock, and small-scale gardening purposes.

Typical Uses

Domestic Use. A properly managed sand-abstraction system provides a suitable source of water for domestic use. A variety of schemes have been developed ranging from small-scale, individual families using basic hand pumps to reticulated systems for small towns.

Irrigation Schemes. These range from small family gardens to large-scale commercial schemes. Where gardens can be established on low river banks (above the flood line), sand-abstraction systems provide an excellent, low-cost option for conveying water to nutrition gardens. One



Figure 1. Typical sand river.



Figure 2. Traditional sand well.



Figure 3. A small-scale community garden where water is drawn by hand pumps from the sand river in the background.

small hand pump (for instance, a Rower) is sufficient to irrigate adequately more than 200 m² of garden—an average size brushwood-fenced garden (Fig. 3).

Livestock Watering. Sand-abstraction systems provide a very suitable source of water for livestock, even in less than optimum conditions. A hand pump, either driven

directly into the sand, discharging into a portable trough on the river sand, or a more complete system drawing water onto the river bank are both appropriate livestock watering systems.

Livelihood. Sand abstraction can provide water for small-scale, rural industry such as brickmaking, household construction and repair, and small livestock projects.

ADVANTAGES AND DISADVANTAGES

Advantages

1. Clean, naturally filtered water free of the contamination associated with open surface water such as dams.
2. The water has a natural taste free from mineral salt contamination often associated with groundwater.
3. Easy to access when compared with the labor involved in building a dam or digging a well.
4. On the traditional and small-scale level, the technology does not require pumps and expensive drilling equipment. It is a low-level technology that people can operate for themselves without outside intervention.
5. Lower cost per cubic meter yield of water compared with ground- or surface water developments.
6. Where extensive sediment deposits exist, large volumes of water can be held.
7. Reduced evaporation from water retained in the sediment as opposed to open surface water.
8. Low environmental impact from installation of sand-abstraction systems.

9. The technology can be used in various other medias such as tube wells and gravel beds.

Disadvantages

1. The system is as vulnerable as any other supply to inadequate recharge from poor rainfall and to seasonal water losses.
2. As there is no confined aquifer at the point of abstraction, if sites are not carefully selected, water can drain away from the site.
3. If not adequately secured, equipment can be washed away in flash floods.
4. The system is based on environmental degradation, it works only where there has been severe soil erosion.
5. At some sites, silt layers form and move through the river sand to clog installations.
6. In certain conditions, infiltration galleries and well points may be subject to bio-fouling.

EPHEMERAL RIVERS

In ephemeral river systems, people have, for generations, been drawing water from river pools and from the river sand. The system works best in large, slow-flowing rivers that cross extensive plains away from the watershed. Conditions for sand abstraction are ideal where rivers cut through igneous soils. Soils in these areas tend to contain large proportions of coarse-grained sands that are carried into and remain in the river systems and allow large volumes of water to be retained. The system is dependent to some degree on continuing soil erosion and is inappropriate in small fast-flowing rivers below the watershed that contain small amounts of unstable sand.

Table 1. Comparison of Community Water Supply Quality—Groundwater and Sand-Abstraction Systems—Zimbabwe

	Recommended Limit	Max Allowed Limit	Source Borehole 1	Source Borehole 2	Source Borehole 3	Source Open Sand Abst	Source 'Safe' Sand Abst
Conductivity (mS/m)	70	300	460	400	940	50	40
pH	6–9	5.5–9.5	7.3	7.5	8.1	7.9	7.3
Turbidity (NTU)	1.0	5.0	30	120	55	20	6.5
Total hardness	20–230	650	1100	110	370	130	120
Calcium (CaCO ₃)	NS	NS	780	20	100	120	68
Magnesium (CaCO ₃)	100	150	320	90	270	10	52
Sodium (Na)	100	400	390	600	910	20	9.7
Potassium (K)	NS	NS	6.5	4.0	5.8	1.5	1.5
Iron (Fe)	0.1	1.0	6.6	11.0	0.9	0.5	1.1
Manganese	0.05	1.0	2.6	0.4	0.2	0.1	2.8
Alkalinity (CaCO ₃)	NS	NS	610	190	500	360	200
Chloride	250	600	22	110	58	2	8
Sulfate	200	600	1600	1700	290	1.7	<0.01
Phosphate	NS	NS	0.1	<0.01	<0.01	<0.01	<0.01
Ammonia total nitrate nitrogen	6.0	10.0	0.3	0.1	0.5	<0.01	<0.01
	–	–	<0.01	<0.01	0.5	<0.01	<0.01
Fluoride	1.0	1.5	0.9	<0.01	<0.01	<0.01	<0.01
Approx dissolved salts	500	1500	2300	2000	4700	250	200
	(WHO)	(WHO)					
Oxygen absorbed 4 h, 27 °C	NS	NS	2.0	<0.01	8.0	3.3	<0.01

^aNS: not significant.

^bWHO

The system is conditional on water freely percolating through sediment to the point of abstraction. Silt and clay create an impenetrable barrier to an abstraction system, and thus sand abstraction in such conditions is not an appropriate option. The ideal situation is a slow-flowing, wide river that has deep, coarse sediment. The ideal site is above a natural rock barrier or in a depression of the riverbed (typically, a former pool), which is continually recharged by water percolating from an expanse of sand above.

The system can be made to work on smaller rivers that contain coarse-grained sand by constructing a subsurface dam to retain sediment. This consists of a barrier to retain the water in the sediment. It is constructed across the river from the riverbed to the surface of the river sediment. A further possibility is a sand dam that is an impoundment, generally a weir, constructed to raise the level of river sediment. The impoundment is designed to silt up over several years; the entire length of the wall is raised by only some 0.3 to 0.5 meters a year. Coarse-grained sand is impounded and the finer silt is washed out of the weir each year. Water can be abstracted from these dams as through any other sand-abstraction system.

GEOGRAPHICAL CONTEXT

Suitable ephemeral rivers are extensive in much of southern Africa, parts of east Africa, the Sahel, the Middle East, the Indian subcontinent, the Southeast United States, parts of Mexico, and parts of Latin America and Australia. The technology of sand abstraction is particularly used in these arid and semiarid areas:

- southern Africa—South Africa, Namibia, Swaziland, Botswana, Zimbabwe, and Zambia;
- east Africa—Kenya, Ethiopia, Sudan, and Somalia.

WATER QUALITY

Unlike groundwater supplies, where water may be contaminated with mineral salts, water obtained from sand-abstraction sources is generally not tainted. Surface water supplies are very quickly contaminated by livestock that wade, urinate, and defecate in the water. Surface water should, therefore, not be considered fit for human consumption without treatment. Where extensive sands exist, the entire waterway acts as an enormous sand filter bed for a sand-abstraction system. Water drawn from a properly constructed scheme may thus be considered potable and require no treatment. However, unprotected sites can become clogged with deposits of clay or fouled with livestock droppings.

SUSTAINABILITY—VILLAGE-LEVEL OPERATION AND MAINTENANCE

1. Sand-abstraction systems draw water from river sands, and thus they are typically shallow water sources requiring only simple hand pumps, which communities that have few tools and little expertise

can operate and maintain. In their simplest form, while still ensuring a protected water supply, buckets are quite adequate for drawing water.

2. Sand-abstraction sites can be identified by villagers, whereas groundwater sources generally require either a geophysical survey or dowsing. A greater sense of ownership and responsibility is thus promoted by sand-abstraction schemes.
3. Many groundwater sources are deep and beyond the capacity of village people to operate and maintain. Boreholes are expensive to drill and can seldom be undertaken by rural communities on their own. In many areas, the aquifers are small and are quickly overabstracted, resulting in blocked fissures and ultimately dry boreholes or wells. Boreholes and wells in sandstone localities are prone to collapse or become completely blocked by fine sediment that slowly percolates into the borehole and granite areas.
4. Small-scale units that can be managed by a Village Water Committee are a very practical option, especially in the Southern Hemisphere, considering the large number of possible sand-filled river sites.
5. In some rivers, there is a possibility that flooding will wash the well point (abstraction point) away. A further complication is that silt layers can move through the river sand and clog installations. Only practical experience can rule out these possibilities.

METHODS OF ABSTRACTING WATER FROM SAND

There are several systems of sand abstraction that have been developed over the years. Each depends on equipment that can be installed into the water-bearing river sand and at all times remain in free moving water. Schemes range in size from small-scale, hand-operated supplies right through to large-scale mechanized (diesel or electric-powered) irrigation schemes on large rivers (Figs. 4 and 5).

Well Screens

Well points or well screens, whose slots are generally 0.5, 1.0, or 1.5 mm wide, prevent entry of river sand into the



Figure 4. Small-scale, hand pump sand-abstraction scheme.



Figure 5. Submersible pumps drawing water for a commercial irrigation scheme from the sand of the Limpopo River in the background.

pump system. Slot size is selected in accordance with the coarseness of the river sand so that sand whose grain size is a diameter greater than the slot is prevented from entering. Smaller sand is initially drawn through the slots but coarser sand quickly collects around the immediate screen and then blocks the entry of finer sand; thus a natural graded filter is developed around the well point.

One or more well screens are connected to a suction pump on the river bank. Installations are done either when the river water is at its lowest level so that the well screen can be dug into the sand, or when the river sand is saturated with water to full depth. Well screens can then be pushed, or 'jetted,' into the lower levels of sand in an artificial 'quicksand' condition, caused by a water jet from a motorized centrifugal pump. This system of installation is relatively technically complex and depends on materials and equipment not readily available (Fig. 6).

Manifold and Well Points

This system depends on low velocity of water at the point of abstraction, sufficient to draw off water without drawing sand into the system. It is best installed toward the end of the dry season when the river water level is at its lowest. River sand is removed to the water-bearing level and a



Figure 6. Well points suitable for small-scale sand-abstraction system.

'manifold' (a large diameter pipe) is laid on the water-bearing sand. Well points are installed at an angle into the sand and connected alternately at each side of the manifold. The manifold is connected to a suction pump on the riverbank. The number of well points, manifold size, and piping to the riverbank are calculated according to the velocity of water required in different parts of the system and on the quantity of water required. This is probably the most common system presently used, and, in general, all necessary materials are readily available.

Infiltration Galleries

Many original sand-abstraction installations were gravity supply systems and consisted of a number of interconnected well points. Sand had to be kept out of the pipes so that water could run in and flow to a sump or false well on the riverbank from where it could be pumped out. Modern synthetic materials have recently improved the potential of this system; before they were introduced, it was difficult to keep sand out. However, it is still difficult to ensure that the pipe system is sufficiently deep in the sand to keep it in water year-round. Further, it necessitates a lot of digging into the riverbank. It is relatively inexpensive and does not require complex equipment or expertise to operate or maintain. A windlass or basic hand pump can be used to draw water.

Caisson

Water is abstracted through the slotted base of a large (1.0 to 1.5 m diameter) flat cylindrical or slightly conical 'caisson' connected by pipes to a suction pump on the riverbank. The 'caisson' is dug into river sand as deeply as possible, and as the water level drops during the season, it is lowered to keep it in water continually. As in any of the systems, the installation is complete once a level has been reached that remains in water year-round. This system is more awkward due to the continual redigging and lowering, but once at its lowest point, can be used where silt tends to accumulate because of the large surface area for abstraction. However, slow recharge through the silt to the abstraction zone can significantly impede abstraction.

COST-EFFECTIVENESS

Small-Scale Schemes

Small sand-abstraction installations do not require any complex machinery such as tractors, graders, drilling rigs, or welders. People at the village level can undertake the installations themselves using primarily locally available equipment. Well points can be easily fabricated in PVC or steel and either jetted, driven, or dug into the saturated sediment. Water can then be abstracted with basic low-tech pumps that do not require expensive operation and maintenance.

Commercial Schemes

Some heavy machinery is needed for initial preparation of the river. However, overall, there is less material to be removed when installing sand-abstraction well points or infiltration galleries than in the constructing an earth embankment dam. Pumps that would typically be used to draw water from a dam or aquifer-based irrigation scheme may also be used in systems based on sand abstraction.

SEDIMENT TRANSPORT

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SEDIMENT SOURCE AREAS

Sediment transport is a natural process of transporting solid particles called sediments from erosion sources to deposition or sedimentation areas. This process has been shaping landscapes throughout geological time. Sediments, sedimentary rocks, and the process by which they were formed are scientifically studied by sedimentology.

One important source of sediments is soil erosion. Geologic soil erosion occurs as surface removal of farm, forest, and other erodible soils in the form of sheet erosion, interrill and rill erosion, gully erosion or deflation, mainly caused by impacts of raindrops (splash erosion, rain

erosion) and overland flow on bare soils or land without dense vegetative cover to protect soils. It may be checked on farm soils by such farm management practices as contour plowing, crop rotation, and mulching. Accelerated erosion is the speeding up of erosive processes, such as deforestation, overgrazing, and construction, either directly or indirectly, by the intervention of humans. Its most obvious effects can be seen in the alteration of a river's regime by improper land use or by human intervention into the catchment area. Other important sediment sources are deposits of mass wasting phenomena, driven by gravity, such as rockfalls, rockslides, landslides, debris flows, and mudflows, exposed to fluvial processes of erosion, transportation, and sedimentation. The subjects of these processes are fluvial sediments—solid particles and particles of rock material that have been transported from their sources and deposited elsewhere by flowing water. A sediment particle is a solid particle, which settles after being suspended in a fluid, but the term usually includes all detritus produced by erosion processes and deposited by fluvial, lacustrine, marine, glacial, and aeolian agencies in the process of sedimentation. The term fluvial sediment is usually reserved for unconsolidated material found in a fluvial system.

PHYSICAL PROPERTIES

Sediment transport involves the two-phase flow of a water–sediment mixture. The most important physical properties of water as a nearly incompressible fluid are the mass density of the fluid (mass of fluid per unit volume), the specific weight of the fluid (fluid weight per unit volume of fluid), the dynamic viscosity (proportional factor between the shear stress causing fluid deformation and the deformation rate), and the kinematic viscosity (the dynamic viscosity of a fluid divided by the mass density of the same fluid). Both the density and the viscosity of water decrease as the temperature increases.

The most important physical properties of sediments as a solid phase can be divided into

- the properties of a single sediment particle: the mass density of solid particles (the mass per unit volume), the specific weight of solid particles (the solid weight per unit volume of solid), the submerged specific weight of a particle (the difference between the specific weights of solid particles and water), the specific gravity (the dimensionless ratio of the specific weight of a solid particle to the specific weight of a fluid at a standard reference temperature), the sediment size (e.g., sieve diameter, fall diameter, nominal diameter), the sediment shape (different shape factors and indexes, e.g., sphericity), and the sediment roundness (different indexes using the sharpness of the edges or corners of a particle to classify particles from angular to rounded);
- the bulk properties of sediments: the particle size frequency distribution (measured by wet or dry sieving equipment such as mechanical shakers or sedimentation tubes and represented as a

histogram or a cumulative curve) and corresponding particle size parameters (mean, median, mode, sorting, skewness, kurtosis), the angle of repose (usually given as the slope angle of a cone of submerged loose material under incipient sliding conditions), the porosity (the volume of voids per total volume), the void ratio (the volume of voids per volume of solid), the unit weight (the weight of solid and water in voids per unit volume), and the dry unit weight (the weight of solid per unit volume).

The most important physical properties of a water-sediment mixture are the specific weight of the mixture (the total weight of solid and water in voids per unit volume), the specific mass of a mixture (the total mass of solid and water in voids per unit volume), the volumetric sediment concentration (the volume of solids divided by the total volume), the dynamic viscosity of a mixture (a function of the dynamic viscosity of water and the volumetric sediment concentration), and the kinematic viscosity of a mixture (the dynamic viscosity of a mixture divided by the mass density of the same mixture).

INCIPIENT MOTION

Due to the stochastic nature of sediment transport, it is difficult to define precisely the flow condition at which a sediment particle will begin to move. Fluid flow around sediment particles resting on the bottom of the flow exerts forces that tend to initiate motion. The resisting force of noncohesive sediment relates to submerged particle weight. Threshold conditions occur when the hydrodynamic overturning moment of forces acting on a single particle exceeds the resisting moment. Then the particle is in incipient motion, thus defining the threshold condition for sediment transport. The hydrodynamic forces acting on a noncohesive sediment particle are the submerged particle weight, the lift force, the drag force, and the resistance force. Most incipient motion criteria are derived from either the shear stress or the velocity approach. Because of the stochastic nature of sediment transport, probabilistic approaches have also been introduced.

The well-known Shields shear stress approach determines the threshold condition from the shear stress, the submerged specific weight of the particle, the particle diameter, the kinematic viscosity, and gravitational acceleration. These factors are grouped into two dimensionless quantities: the dimensionless shear stress called the Shields parameter (relating the shear stress and the submerged weight of a sediment particle), and the shear velocity Reynolds number (expressing the ratio of sediment size to the laminar sublayer thickness). The threshold of motion for granular material of uniform size depends on whether laminar or turbulent flow conditions prevail around the sediment particle; it is given in the form of a Shields diagram showing an experimentally determined relationship between both dimensionless quantities at the threshold condition. The value of the critical Shields parameter depends on whether the boundary is in a hydraulically smooth regime, a transition regime,

or a completely rough regime. Typically, gravel-bed rivers fall within the completely rough regime, and sand-bed rivers fall within the transition and partially hydraulically smooth regime. Shear velocity appears in both the Shields parameter and the shear velocity Reynolds number, so it is possible to define the Shields diagram using the dimensionless particle diameter as an abscissa. For hydraulically rough turbulent flows, the critical shear stress becomes linearly proportional to sediment size.

A velocity approach equivalent to the Shields approach defines the relationship between critical flow velocities and the mean sediment size in the form of a diagram (e.g., Hjølstrom diagram) or defines the relationship between dimensionless critical average flow velocity (expressing the ratio between the average critical flow velocity at incipient motion and the sediment fall velocity in clear water) and the shear velocity Reynolds number (Yang criterion).

The probabilistic approach to incipient motion (proposed by Gessler), gives the mean condition that there is a 50% chance for a given sediment to move under specified flow conditions (turbulence fluctuations) and sediment conditions (particle position, orientation, and protrusion into the flow with respect to particles of different sizes and to bed forms).

MOVEMENT OF FLUVIAL SEDIMENTS

When flow conditions exceed the criteria for incipient motion, sediment particles on a streambed start to move. Once a sediment particle in a stream is in motion, the applied hydraulic load governs the mode of its motion. Generally, there are two modes of motion: near or on the streambed and in suspension.

Sediment that moves near or on the streambed is called bed load transport. Bed load can sometimes be divided into contact load, composed of particles rolling, sliding, or pushing, and saltating load, composed of particles bouncing, hopping, or jumping. The size of the particles that can be transported by saltation is usually correlated with the flow velocity and water density. The bed load, the amount of solid material carried on or near a streambed, usually amounts to less than 10% of the total sediment transported in large alluvial rivers. In mountain rivers and steep high-gradient streams, the major part of the total load may actually be bed load due to numerous mass-movement processes in the watershed and in-channel erosion of deposited fluvial sediments, such as in pools. Under fully developed sediment transport conditions, there is a steady exchange between bed load and suspended load.

Suspended load refers to sediment that stays in suspension for an appreciable length of time. In most natural streams, sediments are mainly transported as suspended load. Discharge-weighted total suspended solids concentrations range from practically 0 mg L^{-1} (clear water) during low flows in mountain environments to several kg L^{-1} during high flows in the form of hyperconcentrated flows, which typically occur in tropical regions and in loess or volcanic ashes.

Based on the modes of sediment transport, the total load is the sum of the bed load and the suspended load.

Based on the source of material transported, the total load can also be classified as the sum of the bed-material load and the wash load. The wash load consists of fine materials that are finer than those found in the streambed. The amount of wash load depends mainly on the supply from the watershed, not on the hydraulics of a stream. Consequently, it is difficult to predict the wash load based on the hydraulic characteristics of a stream.

An important part of a lowland stream's load may also be a dissolved load carried in solution. The proportion varies according to the climate, the chemical nature of the rocks, and the proportion of runoff contributing to the stream flow in relation to the amount of groundwater flow. The quantity of dissolved solids is given by the total concentration of dissolved material in water, measured by complete evaporation of a given quantity of water. The parameter is of use in any examination of the rate of chemical denudation and of the dissolved load of streams. Discharge-weighted total dissolved solids concentrations range from 5000 mg L^{-1} in arid saline environments to 5 mg L^{-1} in tropical rainforest regions.

TRANSPORT CAPACITY OF STREAMS

The ability of a stream to transport sediment, as measured by the maximum quantity (mass or volume) of sediment that can be carried past a specific point in the stream in a given unit of time is called transport capacity. The transport capacity increases as the water discharge becomes greater or the stream gradient becomes steeper, and decreases, as the particle size of the sediment becomes larger. Stream transport capacity is a function of bed width; for a given water discharge and gradient, the flow velocity near the streambed is lower in a wide, shallow stream than in a narrow, deep one.

The sediment discharge ratio in a stream is the ratio between the sediment discharge and the water discharge. When defining sediment discharge, the total solid transport in a stream through its cross section is taken regardless of the mode of sediment motion. The sediment-rating curve is then an empirical expression of the relationship between the stream water discharge and the stream sediment discharge at a given point. It is shown as an equation where the sediment discharge is defined as the water discharge multiplied by the mean sediment concentration and can then be written as a potential function of water discharge.

Flow competence is a term that is used in fluvial geomorphology and hydrology to indicate the ability of a stream to move particles of a particular size as bed load. It refers to the largest particle size that can be carried by a particular stream velocity. The largest particle that can be transported increases generally as a high (e.g., sixth) power of the stream velocity. The stream flow velocity at the bed is controlled by such factors as stream gradient and hydraulic radius. At equal water depth and gradient, near-bed flow velocity is higher in a wide shallow stream, where the hydraulic radius is practically the same as the average water depth, than in a narrow deep one, where, on the contrary, the hydraulic radius is much less than the average water depth. But a large, slowly moving stream

may carry a large quantity of small particles in suspension, and although its (suspended) transport capacity is high, its flow competence is small. Conversely, a small but rapidly flowing stream can move relatively large particles (i.e., it has a high flow competence) although its transport (bed load and suspended load) capacity is small because of its limited water discharge.

Sediment yield is defined as the mean sediment load (total sediment discharge) carried by a stream per unit area of the catchment area; it gives some measure of the rate of erosion in a drainage basin in addition to the transport capacity of the stream itself. To establish the sediment budget of a catchment area, knowledge of sediment production in sediment source areas as well as sediment yield and sediment transport capacity of streams is important.

SEDIMENT TRANSPORT FUNCTIONS

Sediment transport is the mass transfer of solids by flowing water. Its mathematical representation in the form of a transport function needs the determination of sediment transport rates, given as the mass or weight of solids (sediment particles) transported through a cross section of a stream per unit time. Specific sediment transport rates are often given only per unit bed width to be then multiplied by the active streambed width.

Some of the classical formulas for estimating sediment transport rates were derived mainly for bed load, neglecting suspended load. In the way they approach the relation between water discharge and bed load discharge, these bed load formulas can be divided into groups that use shear stresses, energy slope, water discharge, bed forms, probability—turbulent flow fluctuations, stochastic modeling of single particle step lengths and rest periods, regression of laboratory data, and the equal mobility hypothesis as a basis for estimating bed load rates.

The starting point for suspended load equations is the vertical distribution of time-averaged velocities and sediment concentrations at given hydraulic conditions. It can be obtained from the exchange theory that states that under steady-state equilibrium conditions, the downward flux of sediment due to the fall velocity must be balanced by the net upward flux of sediment due to turbulent fluctuations. To obtain the suspended load in many such transport functions, one should know the suspended sediment concentration at some distance above the streambed.

Most total load equations are actually total bed-material load equations. In the comparison between computed and measured total bed-material load, wash load should be subtracted from the measurement before the comparison in most cases. There are two general approaches to determining total load. The first is to compute bed load and suspended load separately and then add them together to obtain total load. The second is to determine the total load function directly without dividing it into bed load and suspended load. A sediment particle may be transported as bed load at one time and as suspended load at another time or location. Except for coarse materials, which are mainly transported as

bed load, total bed-material load equations should be used for determining sediment transport capacity in natural streams.

STREAM AGGRADATION AND DEGRADATION

Sediment transport is a process that interrelates erosion and sedimentation. When the rate of sediment supply from upstream is higher than a stream's sediment transport capacity, the streambed will start to aggrade at a rate defined by the difference between the rate of sediment supply and the sediment transport rate of the stream. The flow in the stream will be saturated with sediment, its transport capacity fully used. But if a stream's sediment transport capacity exceeds the rate of sediment supply from upstream, the balance of sediment load has to come from the channel itself. In this case, the channel starts to degrade.

The particle size distribution of the bed material on a streambed generally depends on the magnitude of the applied hydraulic loads in relation to the mobility of sediment particles of different sizes. In natural gravel-bed and sand-bed rivers, size nonuniformity of fluvial sediments is rather a general rule. The applied hydraulic load expressed as the applied shear stress can move none of the particles (zero sediment transport, no motion), finer and more exposed particles only (weak sediment transport, partial motion), or all particles (fully developed sediment transport, full motion). It is normally observed that without motion of finer particles, the streambed surface is composed of the original bed material. Due to motion of finer particles at a hydraulic load not sufficiently high to displace coarser particles and because of the nonuniformity of the bed-material size, finer material is transported at a faster rate than coarser material, and the remaining bed material on the bed surface becomes coarser. The bed load fractions are finer than those of the bed surface. This stage of the sediment transport is called the selective transport phase. The transition between the zero sediment transport stage and the weak sediment transport stage is smooth. This coarsening process (vertical sorting) on the streambed stops once a layer of coarse material completely covers the streambed and protects the finer materials beneath it from being transported. After this process is completed, the streambed is armored; the coarser layer is called an armor layer. Due to the variation in the flow condition of a natural stream (flood waves), usually more than one layer of armoring material is required to protect finer material beneath it from being eroded. When the applied hydraulic load is large enough to break the coarse armor layer, the finer particles are no longer shielded by stable coarser particles. All available particles of all sizes enter into motion and are found as bed load, as soon as the applied hydraulic load exceeds the threshold of motion of coarser particles. This concept is referred to as the equal-mobility concept for which all fractions of sediment enter into motion at the same value of applied shear stress. When the armor layer breaks, the transition between the weak sediment transport stage and the fully developed sediment transport stage is rather sharp.

The armoring process can be combined with incipient motion criteria to compute the extent of stream degradation. When there is not enough coarse material to develop an armor layer, degradation can be computed using the stable slope approach. This method computes the volume of eroded material in a given stream reach due to limited sediment supply. The limiting slope or final stable slope of the stream can be computed using a sediment transport equation. When there is downstream bedrock or other type of fixed control in the stream, the limiting slope will start at that point and extend in the upstream direction.

DOWNSTREAM FINING IN STREAMS

Sediment transport is definitely selective in many ways and thus is interrelated to sorting processes. Generally, sorting is defined as a process by which materials are graded according to one of their particular attributes, such as shape, size, and density. In sedimentology, the natural sorting of sediments by particle size is a basis for classification into well sorted or poorly sorted deposits. Sediments that have been repeatedly reworked by marine waves or have traveled a long distance downstream in a fluvial system are usually well sorted into different-sized particles, because different particle sizes have different settling velocities. In general, sediments transported by glaciers and mass movement are poorly sorted. Fluvial sediments tend to be well sorted.

Sorting processes in a stream can be observed in three directions: laterally (e.g., differences between coarser sediments in the thalweg and finer sediments on a bar), vertically (e.g., differences between coarser armor layer and finer sublayers), and longitudinally (downstream decrease of mean sediment size due to decreasing flow competence).

The abrasion of fluvial sediments covers attrition processes during passive and active phases of sediment transport, such as breakage, chipping, wearing, and grinding. Consequently, fluvial sediments change their particle size distribution as sediment particles are size-dependently reduced in size and their roundness generally increases. Only breakage, more common in steep high-gradient streams or bedrock reaches, will increase the angularity of sediment particles.

Downstream fining is a process in natural streams that describes an approximately exponential decrease in mean sediment size under ideal conditions. It covers sorting processes and fluvial abrasion, as well as *in situ* processes such as chemical or frost weathering of temporarily stored but exposed fluvial sediments, for example, on a bar. Generally, sorting is the prevailing process responsible for fining; especially in specific situations as when a gravel-bed river suddenly turns into a sand-bed river. Abrasion generally prevails in steep high-gradient streams, where flow competence is high and sorting is thus less pronounced. The exponential decrease is often disrupted in sedimentary links, where fresh sediments from tributaries flow laterally into a stream. When sediment transport over longer reaches of a stream is to be modeled, sorting processes and fluvial abrasion of sediment particles should therefore be accounted for.

MODELING OF SEDIMENT TRANSPORT

On one hand, the problem can be studied practically in the field by observing and measuring most relevant process parameters or in a laboratory using physical (hydraulic) modeling. On the other hand, it can be studied theoretically using numerical modeling and also using appropriate field and laboratory data for model validation.

Sediment transport can be thus defined as a natural process manifesting in sediment-laden flows, and it is usually studied as part of applied mechanics (e.g., river mechanics) or river (loose boundary) hydraulics. The main problem to be solved is to establish functional relationships that relate hydrodynamics (water movement) to sediment movement or vice versa.

For sediment-laden flow, three types of movement can be differentiated:

- the mixture may be considered a Newtonian fluid (the volumetric concentration of solids is very small, e.g., less than 1%)—typically sediment transport in streams and rivers falls into this category;
- the mixture behaves as a quasi-Newtonian fluid (the volumetric concentration of solids remains small, e.g., less than 8–10%)—typically turbidity currents and concentrated suspension fall into this category;
- the mixture behaves as a non-Newtonian fluid (the volumetric concentration of solids becomes important, e.g., more than 8–10%)—typically hyperconcentrated flows (turbidity currents and suspensions) and debris flow fall into this category.

At low sediment concentration and low deformation rates, sediment-laden flow obeys Newton's law of deformation. The governing equations of motion are the continuity and the dynamic equations. These equations can be given in different coordinate systems, such as Cartesian, cylindrical, or spherical, in one, two, or three dimensions.

Modern mathematical models are two- or three-dimensional models, which can be divided into two groups: coupled and uncoupled models. Coupled models solve the governing equations for continuity (mass-balance) and dynamics (momentum-balance) of two-phase flow simultaneously; uncoupled models solve hydrodynamic equations separately from the sedimentologic equations. An important advantage of coupled models is that they can handle intensive unsteady-state processes, such as river changes during a dam break or flash flood waves. Sediment transport numerical models can predict morphological changes and help river engineers to take appropriate measures. Contemporary numerical models use computer power extensively to account for nonuniformity of fluvial sediments when developing fractional models rather than using different mean diameters of sediment.

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STOCHASTIC SIMULATION OF HYDROSYSTEMS

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INTRODUCTION

Simulation is defined as a technique for imitating the evolution of a real system by studying a model of the system. The model is an abstraction, a simplified and convenient mathematical representation of the actual system, typically coded and run as a computer program. If the model has a stochastic¹ element, then we have *stochastic simulation*. The term stochastic simulation sometimes is used synonymously with the *Monte-Carlo method*.

Stochastic simulation is regarded as mathematical experimentation and is appropriate for complex systems, whose study based on analytical methods is laborious or even impossible. For such systems, stochastic simulation provides an easy means to explore their behavior by answering specific 'what if' questions. Moreover, stochastic simulation can be viewed as a numerical method for solving mathematical problems in several fields such as statistical inference, optimization, integration, and even equation solving. Under certain conditions, stochastic simulation is more powerful than other more common numerical methods (e.g., numerical integration of high-dimensional differential equations).

¹The term *stochastic* is etymologized from the Greek verb *στοχάζομαι* initially meaning to aim, point, or shoot (an arrow) at a target (*στόχος* is a target). Metaphorically, the verb meant to guess or to conjecture (the target). The modern Greek meaning is to imagine, to think, to meditate. It appears that the word *stochastic* is found in English since the seventeenth century, and the obsolete meaning pertains to conjecture. Its use as a scientific term is attributed to the Swiss mathematician Jakob Bernoulli (1654–1705), who in his famous masterpiece *Ars Conjectandi* realized that randomness and uncertainty are important aspects of our world and should be objects of scientific analysis. In its modern sense, *stochastic* can be regarded as synonymous with random or probabilistic, but it is mostly used for processes that indicate a mixture of structure and randomness; the term *stochastic process* was used in 1932 by A. N. Kolmogorov.

Due to their complexity, hydrosystems, including water resource systems, flood management systems, and hydropower systems, are frequently studied using stochastic simulation. A generalized solution procedure for hydrosystems problems, including systems identification, modeling and forecasting, hydrologic design, water resources management, and flood management, is discussed. Emphasis is given on the stochastic representation of hydrologic processes, which have a dominant role in hydrosystems. Peculiarities of hydrologic and other geophysical processes (seasonality, long-term persistence, intermittency, skewness, spatial variability) gave rise to substantial research that resulted in numerous stochastic tools appropriate for applications in hydrosystems. Four examples of such tools are discussed: (1) the multivariate periodic autoregressive model of order 1 [PAR(1)], which reproduces seasonality and skewness but not long-term persistence; (2) a generalized multivariate stationary model that reproduces all kinds of persistence and simultaneously skewness but not seasonality; (3) a combination of the previous two cases in a multivariate disaggregation framework that can respect almost all peculiarities except intermittency; and (4) the Bartlett-Lewis process that is appropriate for modeling rainfall and emphasizes its intermittent character on a fine time scale.

A BRIEF HISTORY

Synthetic stream-flow records were first used early in the twentieth century by Hazen (1) in studies of water supply reliability. Their construction, however, was not based on the theory of stochastic processes, then not developed, but on merging and rescaling observed records of several streams. This early work emphasizes the need for long synthetic records and the importance of simulation in water resources technology. The foundation of stochastic hydrology followed the significant developments in mathematics and physics in the 1940s, as well as the development of computers. Specifically, it followed the establishment of the Monte Carlo method, which was invented by Stanislaw Ulam in 1946. Notably, Ulam conceived the method while playing *solitaire* during convalescence from an illness, in an attempt to estimate the probabilities of success of the plays. As Ulam describes the story in some remarks later published by Eckhardt (2), "After spending a lot of time to estimate them by pure combinatorial calculations, I wondered whether a more practical method than 'abstract thinking' might not be to lay it out say one hundred times and simply observe and count the number of successful plays." Soon after, the method grew to solve neutron diffusion problems by himself and other great mathematicians and physicists in Los Alamos (John von Neumann, Nicholas Metropolis, and Enrico Fermi), and was first implemented on the ENIAC computer (2,3). The 'official' history of the method began in 1949 when a paper was published by Metropolis and Ulam (4).

In the field of water resources, the most significant initial steps were the works by Barnes (5) for generating uncorrelated annual flows at a site from normal distribution, Maass et al. (6) and Thomas and Fiering (7)

for generating flows correlated in time, and Beard (8) and Matalas (9) for generating concurrent flows at several sites.

The classic book on time series analysis by Box and Jenkins (10) also originated from different, more fundamental scientific fields. However, it has subsequently become very popular in stochastic hydrology. Box and Jenkins developed a classification scheme for a large family of time series models. Their classification distinguishes among autoregressive models of order p [AR(p)]; moving average models of order q [MA(q)]; combinations of the two, called autoregressive-moving average [ARMA(p, q)] models; and autoregressive integrated moving average [ARIMA(p, d, q)] models. However, despite a large family, Box-Jenkins models do not fully cover the needs of hydrologic modeling, as they do not comply with some peculiarities of hydrologic and other geophysical processes. This gave rise to substantial research that resulted in numerous stochastic tools appropriate for application to water resources.

UTILITY OF STOCHASTIC SIMULATION IN HYDROSYSTEMS

Due to the significant uncertainties inherent in hydrosystems, among which the major uncertainty is hydrologic uncertainty (related to the unknown future of inflows to hydrosystems), an estimate of a system's reliability is important for its design and operation. The reliability of a system is defined as the probability that a system will perform the required function for a specified period of time under stated conditions (11, p. 434). Reliability is the complement of the probability of failure or risk, the probability that "loading" will exceed "capacity." In many instances, the risk can be estimated by analytical means, so stochastic simulation is not required. For example, in the design of dikes that confine a river's flow, the risk of overtopping dikes can be estimated in a typical probabilistic manner, provided that a long enough record of floods of the river exists (some decades). The estimating procedure includes selecting a probabilistic model (e.g., an extreme value distribution function), fitting the model based on the available record, and estimating the probability that a flood will exceed the discharge capacity of the designed river cross section (the estimate of the latter is a matter of hydraulics). Behind this procedure, there are two implicit assumptions that make the methodology appropriate for this problem:

1. The project under study (the dikes) does not modify the natural flow regime, so that if the project had been constructed many years before, the observed flow record would not be altered. Thus, the assumed probabilistic model, although fitted to past data, is still valid after construction of the project.
2. The quantity (flood discharge) whose exceedance was assumed to be the risk is the same quantity, for which we have observed data. Thus, the probabilistic model that was constructed for this quantity can yield the risk directly.

In many cases, however, these assumptions are not valid. Let us first examine the case where assumption 2 is untrue. For example, we may have available rainfall data, from which we can construct a probabilistic model for extreme rainfall intensity, and wish to estimate the probability of exceeding the flood discharge. In this case, we can use a simple one-to-one mapping (transformation) of rainfall to discharge values (e.g., to adopt the relation known as the rational formula), so that the risk of exceeding a certain discharge level equals the risk of exceeding the corresponding rainfall level. This methodology usually incorporates serious oversimplifications and ignorance of certain factors that affect the actual hydrologic process (e.g., retention and infiltration). A more realistic methodology is to use a more detailed model that transforms a rainfall series (not each isolated value) into a discharge series, also considering all processes involved in this transformation. In this case, we can use simulation to obtain a discharge series.

Assumption 1 can be untrue in many cases as well. For example, constructing a dam will alter the flood regime at the dam and downstream, as the spillway outflow does not equal the natural inflow (attenuation occurs due to temporal flood storage). The construction of a storm sewer will also modify the contributing areas and flow times in the area (in addition, it is impossible to have observed data for the sewer discharge in its design phase). Another typical example is a reservoir (see the entry RELIABILITY CONCEPTS IN RESERVOIR DESIGN), whose storage (a quantity that determines the risk, which is the probability of emptying the reservoir) did not exist before the construction of the reservoir. Obviously, in all these cases where assumption 1 is not valid, assumption 2 is also not valid. Thus, we will proceed as in the previous paragraph where simulation is the most appropriate procedure for obtaining a series of data values for the quantity of interest.

Even in an existing project (e.g., an existing reservoir), where the quantity of interest can be measured directly to obtain a historical record, simulation may be necessary again to assess the impacts of several possible changes in the future that were not experienced in the past. For example, a change in water use (e.g., an increase in water demand) and a change in land use or climate, which alters water availability, calls for simulation to estimate a series for the quantity of interest in the scenario examined.

The previous discussion explains why in most studies of hydrosystems (except cases where both assumptions listed previously are valid), it is necessary to simulate to transform some input time series of initial quantities into some output time series of the final quantities of interest. By grace of the power of computers, the simulation methodology has greatly replaced older methodologies that used simplified one-to-one transformations. But why should simulation be stochastic?

In stochastic simulation, the input time series are no longer the observed records but synthetic time series constructed by an appropriate stochastic model. An observed time series is unique and has a limited length equal to the period of observations. On the contrary, a stochastic model can produce as many time series

as required and of any arbitrary length. The utility of a long time series becomes obvious in steady-state simulations (12, p. 1220), when estimating a low value of the probability of failure (risk). For example, in a problem where the accepted probability of failure is 1% per year, apparently several hundreds of simulated years are needed to detect a few failures. The utility of ensemble time series (as opposed to the unique observed record) becomes obvious in nonsteady-state problems (i.e., in terminating simulations) and in forecast problems in which the initial conditions (present and past values of the processes of interest) are known. In these cases, stochastic simulation offers the possibility of different sample paths of the quantity of interest, instead of having a single value at a time, so that we can estimate expected values and confidence zones.

COMPONENTS AND SOLUTION PROCEDURE OF STOCHASTIC SIMULATION

The components and the steps followed in the stochastic simulation of a hydrosystem are shown in Fig. 1. The entire procedure includes two main model components (marked 1 and 2 in Fig. 1) and two simpler procedures (marked 3 and 4 in Fig. 1). The first model component is the stochastic model of inputs, which produces a vector $\mathbf{X}(\mu, \omega)$ of hydrological inputs (e.g., a time series of rainfall, evaporation, and river flow, depending on the problem studied) to the hydrosystem, where μ is a vector that contains the parameters of hydrologic inputs (all estimated from the available records of observations) and ω denotes a sample path realization of the random variables (i.e., ω can be thought of as representing the randomness in the system, e.g., all random numbers in a simulation run). At a minimal configuration, the vector of parameters μ includes mean values, standard deviations, autocorrelations (at least for lag one), and cross-correlations (for multiple-site models).

The second component is the transformation model which takes the inputs $\mathbf{X}(\mu, \omega)$ and produces the outputs $\mathbf{Z}(\mathbf{X}(\mu, \omega), \lambda)$ (e.g., river flow, if $\mathbf{X}$ is rainfall and evaporation, or reservoir release and storage, if $\mathbf{X}$ is river flow); here the vector λ contains parameters of the transformation model (e.g., parameters that determine the hydrologic cycle in a basin and/or parameters that determine the operation of a specific project such as a reservoir).

The third component is a procedure that takes the outputs $\mathbf{Z}(\mathbf{X}(\mu, \omega), \lambda)$ and determines a sample performance measure $L((\mathbf{Z}(\mathbf{X}(\mu, \omega), \lambda)))$ of the system that corresponds to the sample realization represented by ω . This performance measure depends on the problem examined; for instance, in a flood design problem, it can be the risk of exceeding a specified flood level; in a reservoir design problem, it can be either the risk of emptying a reservoir or the attained release for a stated reliability.

By repeating the execution of these three components using different simulation runs, represented by different ω , we can obtain an ensemble of simulations and a sample of performance measures, from which we can estimate the true (independent of ω) performance measure of the

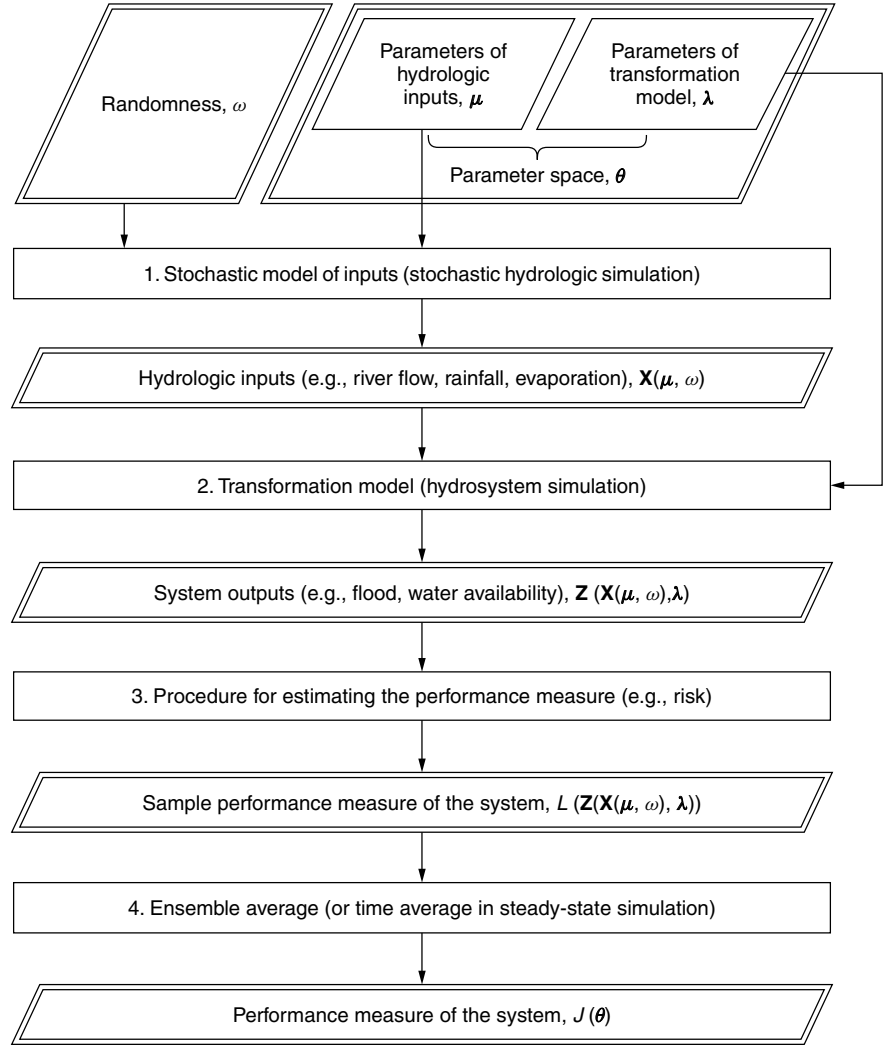


Figure 1. Schematic representation of the components and the solution procedure in hydrosystem simulation. Rectangles represent the components of the solution procedure, and parallelograms represent inputs and outputs of the different components.

system $J(\theta) := E[L(\mathbf{Z}(\mathbf{X}(\mu, \omega), \lambda))]$, where $E[\cdot]$ denotes the expected value and $\theta := (\mu, \lambda)$. However, if the system is stationary and ergodic (in other words, if we have a steady-state simulation), L will tend to J , as the simulation length tends to infinity. Therefore, a single instance of the sample performance measure, estimated from a simulation of large length, is an adequate estimate of the true performance measure. This is the case, for instance, in a reservoir simulation of constant water demand. Conversely, if the water demand is growing in time (a common situation in practice), the simulation is no longer steady-state, and numerous runs, typically of a short length, must be performed to estimate the true performance measure.

In the following part of the article, we will focus on the model component 1, the stochastic model of inputs.

TYPICAL BOX–JENKINS MODELS

Let X_i denote the process of interest (e.g., rainfall or streamflow at a site) where i denotes discrete time. To generate a time series of X_i , we start generating a sequence of independent identically distributed variables (iid, also

known as white noise) V_i that have a specified distribution function (e.g., Gaussian). This is known as generation of random numbers; a concise introduction to this topic can be found in Papoulis (13) and a more detailed presentation can be found in Ripley (14). If it can be assumed that X_i is stationary (i.e., it has a probability distribution function that does not vary in time, which is typically the case if the time step is a year), the unstructured sequence of V_i can be converted to a structured sequence X_i by means of a recursive relationship, whose general form was studied by Box and Jenkins (10). From the large family of Box–Jenkins processes, those that have been widely used in stochastic hydrology are special cases of the equation,

$$X_i = aX_{i-1} + a'X_{i-2} + bV_i + b'V_{i-1} \quad (1)$$

where a, a', b , and b' are parameters that are estimated from the autocovariance properties of the process X_i . The special cases are

1. the iid or white noise or AR(0) process in which $a = a' = b' = 0; b \neq 0$;
2. the Markovian or AR(1) process in which $a' = b' = 0; a, b \neq 0$;

3. the AR(2) process in which $b' = 0; a, b, a' \neq 0$;
4. the ARMA(1, 1) process in which $a' = 0; a, b, b' \neq 0$.

The complete form of (1), known as the ARMA(2, 1) process, has been not used so frequently in hydrology. The special cases listed preserve autocovariance properties for lag 0 (case 1) to lag 2 (cases 2 and 3); beyond these lags, the model autocovariance is zero (case 1) or tends to zero exponentially (cases 2–4).

PECULIARITIES IN STOCHASTIC REPRESENTATION OF HYDROLOGIC PROCESSES

Simple stationary models, such as previously described, are often not the best choice in hydrologic modeling because of several peculiarities of hydrologic processes; the most important are discussed here.

Seasonality. When the timescale of interest is finer than annual, hydrologic processes cannot be regarded as stationary because of the effect of the season of the year on the properties of the process. A simple method often used to take into account seasonality is to standardize the process X_i using seasonal values of mean and standard deviation by setting $Y_i := (X_i - \mu_i)/\sigma_i$ and assuming that Y_i is a stationary process that can be modeled, for instance, by (1); here μ_i and σ_i denote the mean and standard deviation, respectively, of X_i , which, it is assumed, vary with i in a periodic manner. This standardization approach, however, is flawed; the stationarity assumption for Y_i implies that, apart from the mean and standard deviation, other statistical properties of X_i like autocorrelation and skewness do not vary in season, which is not true. A more precise way of modeling seasonality is to assume a cyclostationary (also known as periodic) process, that is expressed as in (1), but has parameters $a, b, a', b', \dots$, and statistics of the noise variables V_i that vary with i in a periodic manner.

Long-Term Persistence. Box–Jenkins models such as (1) are essentially of the short memory type; their autocorrelation structure decreases rapidly with lag time. However, the study of long historical time series of hydrologic and other geophysical processes has revealed that autocorrelations may be significant for large lags of 50 or 100 years. This property is related to the tendency of stream flows to stay above or below their mean for long periods, observed for the first time by Hurst (15), or, equivalently, to multiple timescale fluctuations of hydrologic processes (16). Therefore, models such as (1) are proven inadequate in stochastic hydrology, because it is important that the long-term persistence of hydrologic processes is reproduced (see the entry RELIABILITY CONCEPTS IN RESERVOIR DESIGN).

Intermittency. On fine timescales, some hydrologic processes such as rainfall and in some cases stream flow appear as intermittent processes; thus, rainfall alternates between two states, dry (zero rainfall) and wet (positive rainfall). This is manifested in the marginal probability distribution of rainfall depth by a discontinuity at zero. Box–Jenkins processes such as that in (1) need to be

truncated to represent this discontinuity; this is not so easy, nor common. To model intermittency, alternative two-state processes, such as two-state Markov chains (17, p. 302) and point process models (18) have been proposed.

Skewness. Another peculiarity of hydrologic processes is the skewed distribution functions observed mostly on fine and intermediate timescales. This is not so common in other scientific fields whose processes are typically Gaussian. Therefore, attempts have been made to adapt standard models to enable treatment of skewness (19–22). Skewness is caused mainly by the fact that hydrologic variables are nonnegative and sometimes intermittent. Therefore, successful modeling of skewness indirectly contributes to avoiding negative values of simulated variables; however, it does not eliminate the problem, and some ad hoc techniques (such as truncating negative values) are often used in addition to modeling skewness.

Spatial Variation. Hydrologic processes evolve in both time and space. Typically, time series models consider only temporal evolution. The most precise mathematical representation of hydrologic processes can be achieved by extending the indexing set of the process from one dimension (representing time) to three dimensions (one for time and two for space). However, multidimensional modeling is not easy and has been implemented only in a few cases (for example in continuous time and space modeling of rainfall; 23). A midway solution, which is the most common in stochastic hydrology, is to use multivariate models, that describe the temporal evolution of the process simultaneously at a number of points. The same method can be used directly to model more than one cross-correlated hydrologic process (e.g., rainfall and runoff) at the same location simultaneously.

In the following sections, we give some characteristic examples of models that respect these peculiarities and together can deal with a large spectrum of problems in stochastic hydrologic simulation:

1. The multivariate periodic autoregressive model of order 1 [PAR(1)], which reproduces seasonality and skewness but not long-term persistence;
2. A generalized multivariate stationary model that reproduces all kinds of persistence and simultaneously skewness but not seasonality;
3. A combination of the previous two cases in a multivariate disaggregation framework that can respect almost all the previously listed peculiarities, except intermittency which such types of models may not easily handle;
4. The Bartlett–Lewis process that is appropriate for modeling rainfall and emphasizes its intermittent character on a fine timescale, but only on a single point basis.

The Multivariate PAR(1) Model

Let $\mathbf{X}_s := [X_s^1, X_s^2, \dots, X_s^n]^T$ represent a hydrologic process on a subannual (e.g., monthly) timescale (δ) and at n locations (the subscript T denotes the transpose of a vector

or matrix). The PAR(1) model is similar to the AR(1) model but has periodically varying parameters. In a multivariate setting, it is expressed by

$$\mathbf{X}_s = \mathbf{a}_s \mathbf{X}_{s-1} + \mathbf{b}_s \mathbf{V}_s \quad (2)$$

where $\mathbf{a}_s$ and $\mathbf{b}_s$ are $(n \times n)$ matrices of parameters and $\mathbf{V}_s$ is a vector of innovations (independent, random variables in both time and location) of size n . The time index s can take any integral value, but the parameters $\mathbf{a}_s$ and $\mathbf{b}_s$ are periodic functions of s whose period $k := 1 \text{ year}/\delta$ (e.g., 12 if δ is one month). This model can reproduce the following set of statistics:

1. the mean values, i.e., the k vectors $\boldsymbol{\mu}_s := E[\mathbf{X}_s]$ each of size n ;
2. the variances and lag-zero cross-covariances among different locations, i.e., the k matrices $\boldsymbol{\sigma}_{ss} := \text{Cov}[\mathbf{X}_s, \mathbf{X}_s] = E[(\mathbf{X}_s - \boldsymbol{\mu}_s)(\mathbf{X}_s - \boldsymbol{\mu}_s)^T]$ (where $\text{Cov}[\cdot]$ denotes covariance), each of size $(n \times n)$;
3. the lag-1 autocovariances at each location, i.e., the k vectors $\boldsymbol{\gamma}_{s,1} := [\gamma_{s,1}^1, \dots, \gamma_{s,1}^n]^T$, where $\gamma_{s,\tau}^l := \text{Cov}[X_s^l, X_{s-\tau}^l] = E[(X_s^l - \mu_s^l)(X_{s-\tau}^l - \mu_{s-\tau}^l)]$, each of size n (notice the notational identity $\gamma_{s,0}^l \equiv \sigma_{ss}^l$);
4. the third moments, the k vectors $\boldsymbol{\xi}_s = \mu_3[\mathbf{X}_s] = E[(X_s^l - \mu_s^l)^3, l = 1, \dots, n]^T$ each of size n (where $\mu_3[\cdot]$ denotes the third central moment of a random variable or random vector).

The model parameters $\mathbf{a}_s$ and $\mathbf{b}_s$ are typically determined by the moment estimators that are

$$\mathbf{a}_s = \text{diag}(\gamma_{s,1}^l / \gamma_{s-1,0}^l, l = 1, \dots, n) \quad (3)$$

$$\mathbf{b}_s \mathbf{b}_s^T = \boldsymbol{\sigma}_{ss} - \mathbf{a}_s \boldsymbol{\sigma}_{s-1, s-1} \mathbf{a}_s \quad (4)$$

These equations are extensions of the seasonal model of those for the stationary Markov model given by Matalas and Wallis (19, p. 63). In an alternative estimate, a full (rather than a diagonal) matrix $\mathbf{a}_s$ can be derived, which enables preserving the lag-1 cross-covariances among different locations. However, the more parsimonious formulation in Eq. 3 is sufficient for most cases. The calculation of $\mathbf{b}_s$, given the product $\mathbf{b}_s \mathbf{b}_s^T$ from Eq. 4, is not a trivial issue. A generalized methodology to do this operation, also known as the extraction of the square root of a matrix, was proposed by Koutsoyiannis (21). Another group of model parameters are the moments of the auxiliary variables $\mathbf{V}_s$. The first moments (means) are obtained by

$$E[\mathbf{V}_s] = \mathbf{b}_s^{-1}(\boldsymbol{\mu}_s - \mathbf{a}_s \boldsymbol{\mu}_{s-1}) \quad (5)$$

By definition 1, the variances are $\text{Var}[\mathbf{V}_s] = [1, \dots, 1]^T$, and the third moments are obtained by

$$\mu_3[\mathbf{V}_s] = (\mathbf{b}_s^{(3)})^{-1}(\boldsymbol{\xi}_s - \mathbf{a}_s^{(3)} \boldsymbol{\xi}_{s-1}) \quad (6)$$

where $\mathbf{a}_s^{(3)}$ and $\mathbf{b}_s^{(3)}$ denote the matrices whose elements are the cubes of $\mathbf{a}_s$ and $\mathbf{b}_s$, respectively.

A GENERALIZED MULTIVARIATE STATIONARY MODEL RESPECTING LONG-TERM PERSISTENCE

The most difficult and often the most important task in simulating hydrologic processes on an annual scale is to reproduce long-term persistence. The Box–Jenkins processes are inappropriate for this purpose. Other types of models such as fractional Gaussian noise (FGN) models and broken line models (whose comprehensive discussion can be found in Ref. 24) have several weak points such as parameter estimation problems, narrow type of autocorrelation functions that they can preserve, and their inability to reproduce skewness and simultaneously to perform in multivariate problems. In a recent paper (22), all these problems were remedied; the proposed generalized methodology can perform in multivariate problems for all categories of statistics listed in points 1–4 of the previous section and, in addition, for the autocovariances at all locations for any lag r .

The setting of the method is stationary, rather than cyclostationary, so all statistics and parameters are not functions of time, which is reflected in the notation used. For example, the autocovariance for lag τ is denoted as $\boldsymbol{\gamma}_\tau := [\gamma_\tau^1, \dots, \gamma_\tau^n]^T$ where $\gamma_\tau^l := \text{Cov}[X_i^l, X_{i-\tau}^l]$. Remember that long-term persistence implies nonignorable autocovariances for high lags (e.g., of the order $10^2 - 10^3$). Such autocovariances can be described by a power-type (as opposed to the exponential type of ARMA processes) function such as

$$\gamma_\tau^l = \gamma_0^l (1 + \kappa^l \beta^l \tau)^{-1/\beta^l} \quad (7)$$

where κ^l and β^l are constants. This generalized autocovariance structure (GAS) incorporates as special cases the exponential ARMA type structure (for $\beta = 0$) and the FGN structure (for a special combination of κ^l and β^l ; see Ref. 22). The constants κ^l and β^l can be estimated by fitting Eq. 7 to the sample autocovariance estimates; note that Eq. 7 can be used for lags beyond a certain lag τ_0 , thus allowing the possibility of specifying different values (i.e., the historical values of the sample autocovariance estimates) for smaller lags.

In each of the locations, the process X_i^l can be expressed in terms of some auxiliary variables V_i^l , uncorrelated in time i (i.e., $\text{Cov}[V_i^l, V_m^k] = 0$ if $i \neq m$), but correlated in different locations l for the same time i by using

$$X_i^l = \sum_{r=-q}^q a_{|r|}^l V_{i+r}^l \quad (8)$$

This equation defines the so-called symmetrical moving average (SMA) scheme. Like the conventional (backward) moving average (MA) process, the SMA scheme transforms a sequence of temporally uncorrelated variables V_i^l into a process with autocorrelation by taking the weighted average of a number of V_i^l . In the SMA process, the weights a_r^l are symmetrical about a center (a_0^l) that corresponds to the variable V_i^l . The number of variables V_i^l that define X_i^l is $2q + 1$, where q is theoretically infinity but in practice can be restricted to a finite number, as the sequence of weights a_r^l tends to zero for

increasing r . Koutsoyiannis (22) showed that the discrete Fourier transform $s_a^l(\omega)$ of the a_r^l sequence is related to the power spectrum $s_\gamma^l(\omega)$ of the process (i.e., the discrete Fourier transform of the sequence of γ_r^l) by

$$s_a^l(\omega) = \sqrt{2s_\gamma^l(\omega)} \quad (9)$$

This enables easy and fast (using the fast Fourier transform) computation of the sequence of a_r^l , even if the terms of the sequence are thousands. The computation includes transforming the sequence of γ_r^l to $s_\gamma^l(\omega)$, calculating $s_a^l(\omega)$ from Eq. 9, and inversely transforming $s_a^l(\omega)$ to the sequence of a_r^l .

The auxiliary variables V_i^l , by definition, have unit variances, means $E[V_i^l]$, and third central moments $\mu_3[V_i^l]$ given by

$$\left(a_0 + 2 \sum_{j=1}^s a_j\right) E[V_i^l] = \mu^l, \quad \left(a_0^3 + 2 \sum_{j=1}^q a_j^3\right) \mu_3[V_i^l] = \xi^l \quad (10)$$

Their variance–covariance matrix $\mathbf{c} := \text{Cov}[\mathbf{V}_i, \mathbf{V}_i]$ has elements c^{lk} that can be expressed in terms of σ^{lk} (the elements of the variance–covariance matrix σ of $\mathbf{X}_i$), and the sequences a_r^l and a_r^k by

$$c^{lk} = \sigma^{lk} / \sum_{r=-q}^q a_{|r|}^l a_{|r|}^k \quad (11)$$

Given the matrix $\mathbf{c}$, the vector of variables $\mathbf{V}_i = [V_i^1, V_i^2, \dots, V_i^n]^T$ can be generated using the simple multivariate model,

$$\mathbf{V}_i = \mathbf{b}\mathbf{W}_i \quad (12)$$

where $\mathbf{W}_i = [W_i^1, W_i^2, \dots, W_i^n]^T$ is a vector of innovations with unit variance, independent both in time i and in location $l = 1, \dots, n$, and $\mathbf{b}$ is a matrix of size $n \times n$ such that

$$\mathbf{b}\mathbf{b}^T = \mathbf{c} \quad (13)$$

The other parameters needed to define model Eq. 12 completely are the vector of mean values $E[\mathbf{W}]$ and third moments $\mu_3[\mathbf{W}]$ of W_i^1 . These can be calculated in terms of the corresponding vectors of V_i^1 , already known from Eq. 10, by

$$E[\mathbf{W}] = \mathbf{b}^{-1}E[\mathbf{V}], \quad \mu_3[\mathbf{W}] = (\mathbf{b}^{(3)})^{-1}\mu_3[\mathbf{V}] \quad (14)$$

STOCHASTIC DISAGGREGATION TECHNIQUES

Seasonal models that can reproduce the long-term persistence of hydrologic processes do not exist at present. If the timescale of interest is finer than annual and, simultaneously, respecting of long-term persistence is important, a two-scale approach is followed. A stationary stochastic model like that described in the previous section is used to generate the annual time series. These are then disaggregated into a finer timescale so that the periodicity and short-term memory of the process of interest are respected. Traditionally, the latter task has

been tackled by the so-called disaggregation models, which were initially proposed by Valencia and Schaake (25) and improved since then by the contribution of several researchers (for an outline of such contributions, see Refs. 26 and 27). These are purposely designed models to generate a process at the finer timescale given than at the coarser one. Specifically, they do not model the process of interest in the lower level timescale itself, but rather they are hybrid schemes simultaneously using both timescales. Sometimes (owing to nonlinear transformations of variables) these models cannot ensure consistency with the higher level process. Then, adjusting procedures are necessary to restore consistency (26,28).

A different approach was recently proposed by Koutsoyiannis (29), which is a generalized framework for coupling stochastic models on different timescales. This approach couples two independent stochastic models appropriate, respectively, for the coarser (annual) and finer (e.g., monthly) scales using a transformation that modifies the output of the latter to become consistent with the series produced by the former model. To demonstrate this approach, we will assume that the coarse scale model is the multivariate SMA model described in the previous section, which produces annual series $\mathbf{Z}_i$, and the finer scale model is the multivariate PAR(1) model described two sections before, which produces monthly series $\mathbf{X}_s$. Consistency of the two series requires that they obey

$$\sum_{s=(i-1)k+1}^{ik} \mathbf{X}_s = \mathbf{Z}_i, \quad (15)$$

where k is the number of fine-scale time steps within each coarse-scale time step ($k = 12$ in our example). The annual series $\mathbf{Z}_i$ is generated first. The finer scale model is run independently of the coarser scale one, without any reference to the known $\mathbf{Z}_i$, and produces monthly series $\tilde{\mathbf{X}}_s$. If we aggregate the latter on the annual scale (by means of Eq. 15), we will obtain some annual series $\tilde{\mathbf{Z}}_i$, which will apparently differ from $\mathbf{Z}_i$. In a subsequent step, we modify $\tilde{\mathbf{X}}_s$ thus producing $\mathbf{X}_s$ consistent with $\mathbf{Z}_i$ (in the sense that they obey Eq. 15) without affecting the stochastic structure that characterizes $\tilde{\mathbf{X}}_s$. For this modification, we use a linear transformation $\mathbf{X}_s = \mathbf{f}(\tilde{\mathbf{X}}_s, \tilde{\mathbf{Z}}_i, \tilde{\mathbf{Z}}_i)$ which has been termed the coupling transformation. This is given by (29)

$$\mathbf{X}_i^* = \tilde{\mathbf{X}}_i^* + \mathbf{h}(\mathbf{Z}_i^* - \tilde{\mathbf{Z}}_i^*) \quad (16)$$

where

$$\mathbf{X}_i^* := [\mathbf{X}_{(i-1)k+1}^T, \dots, \mathbf{X}_{ik}^T]^T \quad (17)$$

$$\mathbf{Z}_i^* := [\mathbf{Z}_i^T, \mathbf{Z}_{i+1}^T, \mathbf{X}_{(i-1)k}^T]^T \quad (18)$$

$$\mathbf{h} = \text{Cov}[\mathbf{X}_i^*, \mathbf{Z}_i^*] \{\text{Cov}[\mathbf{Z}_i^*, \mathbf{Z}_i^*]\}^{-1} \quad (19)$$

and $\tilde{\mathbf{X}}_i^*$ and $\tilde{\mathbf{Z}}_i^*$ are defined in terms of $\tilde{\mathbf{X}}_s$ and $\tilde{\mathbf{Z}}_i$ in a manner identical to that of the definition of $\mathbf{X}_i^*$ and $\mathbf{Z}_i^*$.

It is clarified that the vector $\mathbf{X}_i^*$ contains the monthly values of all 12 months of year i for all examined locations (e.g., for five locations, $\mathbf{X}_i^*$ contains $12 \times 5 = 60$ variables) and the vector $\mathbf{Z}_i^*$ contains (1) the annual values of the

current year; (2) the annual values of the next year; and (3) the monthly values of the last month of the previous year (e.g., for five locations, $\mathbf{Z}_i^*$ contains $3 \times 5 = 15$ variables). Items (2) and (3) of $\mathbf{Z}_i^*$ are included to ensure that the transformation will preserve the covariance properties among the monthly values of each year and also the covariances for the previous and next years as well. Note that at the stage of the generation at year i , the monthly values of year $i - 1$ are known (therefore, in $\mathbf{Z}_i^*$, we enter monthly values of the year $i - 1$), but the monthly values of year $i + 1$ are not known (therefore, in $\mathbf{Z}_i^*$, we enter annual values of the year $i + 1$, which are known).

The quantity $\mathbf{h}(\mathbf{Z}_i^* - \tilde{\mathbf{Z}}_i^*)$ in Eq. 16 represents the correction applied to $\tilde{\mathbf{X}}$ to obtain $\mathbf{X}$. Whatever the value of this correction, the coupling transformation will ensure preservation of first- and second-order properties of variables (means and variance-covariance matrix) and linear relationships among them (in our case the additive property, Eq. 15). However, it is desirable to have this correction as small as possible so that the transformation does not seriously affect other properties of the simulated processes (e.g., the skewness). It is possible to make the correction small enough, if we keep repeating the generation process for the variables of each period (rather than performing a only single generation) until a measure of the correction becomes lower than an accepted limit. This measure can be defined as

$$\Delta = (1/m) \|\mathbf{Z}_i^* - \tilde{\mathbf{Z}}_i^*\| \quad (20)$$

where $\mathbf{Z}_i^*$ and $\tilde{\mathbf{Z}}_i^*$ are respectively, $\mathbf{Z}_i^*$ and $\tilde{\mathbf{Z}}_i^*$ standardized by the standard deviation (i.e., $\mathbf{Z}_i^{*sl} := \mathbf{Z}_i^{*l} / \{\text{Var}[\mathbf{Z}_i^{*l}]\}^{1/2}$), m is the common size of $\mathbf{Z}_i^*$, and $\|\cdot\|$ denotes the Euclidian norm.

POINT PROCESS MODELS

On even finer timescales such as daily or hourly, the intermittency of hydrologic processes dominates, and stochastic models like those described earlier can hardly describe it. Point process models have

been the most widespread approach to representing intermittent hydrologic processes, particularly, rainfall. As a representative example, we summarize here the rainfall model based on the Bartlett–Lewis process; this was chosen due to its wide applicability and experience in calibrating and applying it to several climates. Accumulated evidence of its ability to reproduce important features of the rainfall field from the hourly to the daily scale and above can be found in the literature (30–32). This type of model has the important feature of representing rainfall in continuous time; the statistical properties on any discrete timescale are directly obtained from those in continuous time, and this enables model fitting combining statistics of different timescales.

The Bartlett–Lewis rectangular pulse model assumes that rainfall occurs in the form of storms of certain durations and each storm is a cluster of random cells; each storm has constant intensity during the time period it lasts. These are the general assumptions of the model (Fig. 2):

1. Storm origins t_i occur following a Poisson process at rate λ (this means that durations between consecutive storm origins, $t_i - t_{i-1}$, are independent identically distributed following an exponential distribution whose parameter is λ).
2. Origins t_{ij} of cells of each storm i arrive following a Poisson process at rate β .
3. Arrivals of each storm i terminate after a time v_i exponentially distributed with parameter γ .
4. Each cell has a duration w_{ij} exponentially distributed with parameter η .
5. Each cell has a uniform intensity X_{ij} with a specified distribution.

In the original version of the model, it is assumed that all model parameters are constant. In a modified version, parameter η is randomly varied from storm to storm using a gamma distribution of shape parameter α and scale parameter ν . Subsequently, parameters β and γ also vary so that the ratios $\kappa := \beta/\eta$ and $\phi := \gamma/\eta$ are constant.

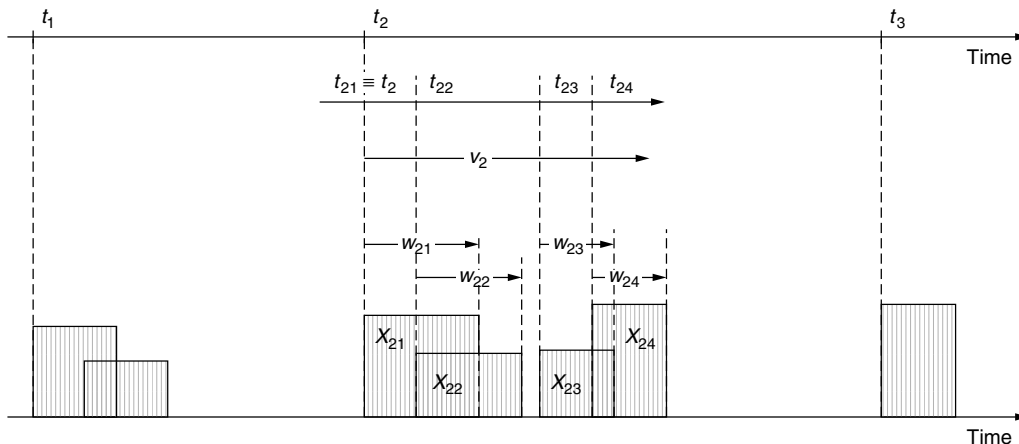


Figure 2. Explanatory sketch for the Bartlett–Lewis rectangular pulse model.

The distribution of the uniform intensity X_{ij} is typically assumed exponential with parameter $1/\mu_X$. Alternatively, it can be chosen as a two-parameter gamma with mean μ_X and standard deviation σ_X . Thus, in its most simplified version, the model uses five parameters, λ , β , γ , η , and μ_X , and in its most enriched version, seven parameters, λ , κ , ϕ , α , ν , μ_X , and σ_X .

The equations of the Bartlett–Lewis model, relating the statistical properties of the rainfall process in discrete time to the model parameters, may be found in the references mentioned before. These equations serve as the basis for model fitting.

CONCLUDING REMARKS

Stochastic simulation is a powerful method, easily applicable, and extremely flexible. Its main advantage is its ability to perform in complex systems describing them faithfully, without simplifying assumptions. However, it is an approximate procedure, and the accuracy of its results depends on the sample size. In addition, it is a slow procedure, as the estimation error decreases inversely proportional to the square root of the simulation length (i.e., for half the error, we need four times greater simulation length). Today, this is not a major problem as the progress in computer technology makes attainable even a vast simulation length in reasonable computer time.

In addition to the estimation error due to a finite simulation length, another significant source of uncertainty is always the limited historical records (based on hydrologic measurements), which are used to fit probabilistic or stochastic models. This source of uncertainty, which concerns the simulation method and also any method, including an analytical one, is forgotten sometimes, so the following points of caution should be stressed:

- The choice of a particular stochastic model and the estimation of its parameters are always based on available historical records, which are the only authentic source of information.
- Simulated (synthetic) hydrologic records do not replace historical records.
- The generation of a synthetic record (whose length is usually a multiple of that of the historical record) does not add any information nor does it extend the historical record length.

In conclusion, the following points should be added:

- In problems that can be solved analytically (as in the example of the design of dikes discussed earlier), stochastic simulation is not the preferable method.
- Stochastic simulation becomes a powerful numerical method when studying a complex system and analytical (or other numerical) methods are not applicable, are very difficult, or require oversimplifying assumptions for the system.

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STORAGE AND DETENTION FACILITIES

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The increase of population in the cities results in urbanization of undeveloped areas, which increases impervious areas, e.g., paved roads, parking facilities, residential houses, and commercial property development. The effects of urbanization include a pronounced impact on the characteristics of an area's hydrology; i.e., urbanization leads to an increase of peak flow rate and a reduction of the time to peak of runoff hydrographs. A common practice in storm water management is that the increase in peak flow rate caused by urbanization must be mitigated by reducing the peak flow rate to attain the predevelopment value by using storage and detention facilities. A storage and detention facility (pond or basin) is a low-lying area (natural or constructed), which is designed to hold a specific amount of runoff temporarily and used to attenuate peak flow rate. Once runoff has been collected in a detention facility, it is released to downstream conveyances (e.g., channels, bayous, or lakes) in a controlled manner such that downstream flooding and other adverse impacts are prevented or alleviated. Detention facilities can be surface and underground storage basins. Underground detention facilities are often used for small commercial development and consist of a series of connected large pipes or prefabricated custom chambers manufactured specifically for underground detention; e.g., a supermarket can install an underground detention pipe system under its parking lot to satisfy the zero runoff increase requirement specified by a

regulatory agency. A surface detention pond is typically much larger in area and used for regional stormwater management, and it is a depressed or excavated area with earth embankments. Figure 1 shows a schematic of inflow hydrograph (after urban development) and outflow hydrograph from the outlet of a detention pond, and the peak flow of outflow hydrograph is equal to or smaller than the peak flow before development, which is much lower than the peak flow in the inflow hydrograph. From Fig. 1, one can see that the required volume of a detention pond is equal to the difference in area under inflow and outflow hydrographs.

Detention ponds can be classified as wet or dry ponds, depending on whether the principle outlet is constructed above the lowest point in the depression. The wet pond, also known as the retention pond, is designed to retain some of the inflow inside the pond. About 80% or more of detention facilities in the United States and Canada are dry ponds during no rain season, and they provide the maximum capacity for runoff detention during the next rainfall event. Sometimes pipes or channels may be necessary to divert inflow into detention facilities if the detention pond is outside of the main drainage pathway (off-line pond). An on-line pond is positioned along the drainage pathway, and all runoff naturally pass through it without any inflow diversion or collection system. Detention storage designs must consider the size of the pond, site selection, outflow structure (weir or orifice) design, construction of pond (cutting and filling), maintenance cost, and estimates of service life. Multiple ponds may be necessary to achieve the detention requirement under certain site conditions, particularly where the topography is flatter.

Design criteria for detention ponds vary widely. In the United States, detention regulations are typically adopted at the municipal or county level. Design of a detention pond like any other hydrologic design is always associated with one or more design storm events, e.g., typically specified as a certain return period. For example, a drainage district is to design a detention pond to hold all runoff for a 300-acre developed area during a 100-year, 24-hour rainfall event. The 100-year, 24-hour rainfall for the study area is 12 inches. If available land for the detention pond is 40 acres, what will be the depth for the pond? The depth will be 7.5 ft ($300 \text{ acre} \times 12 \text{ inches} / 40 \text{ acre} / 12$). This

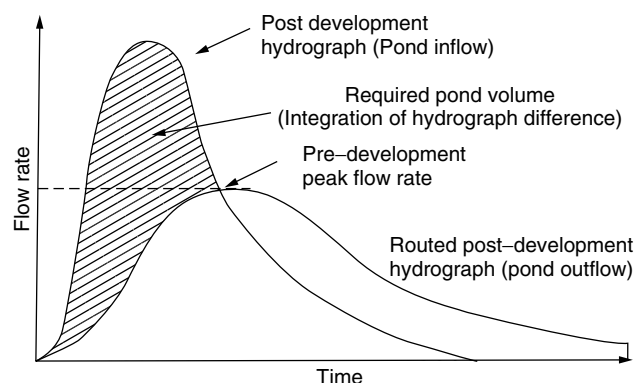


Figure 1. Routing of runoff hydrograph through detention basin.

detention pond design is simple because the pond holds all runoff during the rainfall event and the district plans to manually control outflow-gate to release the flow after the rainfall event. This design does not require any hydrologic routing analysis, but it requires a large amount of land requisition (40 acres).

The design of the storage volume (size) and outlet structure often requires knowledge of the predevelopment peak flow and an estimate of the inflow hydrograph under urbanized conditions. It needs to perform hydrologic routing analysis for the inflow hydrograph into the pond. Hydrograph routing (1) can predict the temporal and spatial variation of an inflow hydrograph as it traverses a river reach or reservoir (pond). Routing through a detention pond is meant to solve the continuity equation by using the storage–outflow relation. The storage–outflow relation is typically determined from two relationships: outflow (discharge) versus water surface elevation and storage (volume) versus water surface elevation. The first relationship is determined by the outflow structure design, and the second one is determined by the detention pond geometry (change horizontal area with respect to elevation) from contour or topographic maps.

Hydraulic analysis for detention pond design determines outflow through the outlet under different water surface elevations inside the pond, and outflow rate depends on type and geometry of the outlet. The most common outlet structures for detention ponds can be of orifice-type, weir-type (broad-crested, sharp-crested, spillways, and v-notch weir), or a combination of the two. The exit flow can be classified as submerged or free and depends on the effective head of water surface on the upstream side of the outlet structure. The outflow from each type of outlet structure is a function of several variables. For the flow through the orifice-type (e.g., a culvert), it is function of orifice area and the difference between the head water elevation and the centroid of the orifice. For the weir-type outlet (emergency spillway), it is function of effective crest length and the head over the weir crest. For the V-notch weir, it is a function of the notch angle and the head over the notch bottom. The riser pipe acts as weir at low heads and like an orifice at higher heads. The equations required to calculate the outflow from the detention pond for different outlet structures can be found in many textbooks and design handbooks (2,3).

A detention pond can be formed in a low-lying area by only constructing the earth embankment with the outflow structure and it could be fully constructed, i.e., digging a pond for certain depth (construction cost is high but sometimes that is the only choice). The side slopes of a constructed pond should be stabilized where needed to prevent the erosion. The side slopes should be 3H:1V or flatter (4). Outlets should have trash racks, and coarse gravel packing should be provided if a perforated riser outlet prevents clogging of the riser caused by accumulation of sediment particles. Criteria for selecting the site for install of a pond should include the site's ability to support the environment as well as the cost effectiveness of locating a pond at that specific site. The pond should be located where the topography of the site allows for maximum storage at minimum construction cost (5). A

site-specific constraint for pond construction includes underlying bedrock that would require expensive cutting operations to excavate. In designing the wet detention pond, the site must have adequate base-flow from the groundwater or from the drainage area to maintain the permanent pool. The storage capacity of the detention pond may be maintained properly by designing the outlet structure properly (appropriate opening size), so that the velocity of the water through the outlet as well as through the pond does not cause erosion of the bed material, interns which may cause clogging of the outlet opening.

The detention pond design process can be summarized as the following steps:

1. Select appropriate site and shape for detention pond design, and select material for detention structure construction.
2. Estimate inflow hydrograph and the peak inflow from watersheds to the detention pond.
3. Estimate the volume of water that can be stored in the detention pond and establish an elevation-storage relationship from the geometry of the pond.
4. Select the proper type of outlet structure and design the outlet opening.
5. Perform hydrologic routing analysis to determine the peak flow rate of the outflow hydrograph, to check whether it is smaller than the allowable one (peak rate before development) or it is safe, which does not cause flooding at downstream. If it does not satisfy the criteria, redesign the outlet or increase the height of the retaining structure to increase the storage.

Estimation of the size of a detention pond can also be accomplished by some simplified methods (1), e.g., natural storage loss method, modified Rational triangular hydrograph method, and NRCS TR-55 procedure, and by using computer software packages. Under normal operation of a detention pond, it is assumed that sufficient storage is available to regulate a flood without overtopping of the pond. Under special circumstances, the emergency operation scheme is implemented when the storage capacity is limited and inflows caused by heavy rainfall events (with a return period longer than the design return period) are expected to exceed the storage capacity (overtopping). Therefore, an emergency spillway should be provided to control overflow relief for large storms. The detention pond can also be designed to improve the water quality by removing some pollutants through sedimentation and other mechanisms as runoff is temporarily stored in the pond before the water is draining to another natural water body.

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URBAN STORMWATER RUNOFF WATER QUALITY ISSUES

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INTRODUCTION

Urban creeks and lakes can provide important habitats for aquatic life, as well as aesthetic assets to communities. A key component of this resource is the quality of water in these waterbodies. This chapter is devoted to a review of water quality problems in urban creeks and lakes caused by stormwater runoff-associated pollutants.

The primary function of many urban creeks is the conveyance of stormwater to prevent flooding. Often they have been channelized to assist in achieving rapid removal of stormwater from an urban area. This channelization, coupled with the development (paving) in the urban creek watershed, is at odds with providing high-quality aquatic life habitat. Urban creek flows can vary from a few cubic feet per second of groundwater-based flow to a thousand or more cubic feet per second during flood flow conditions associated with major runoff events. The high flows are detrimental to developing and maintaining desirable aquatic life habitat. Urban creeks also are frequently receptacles for waste materials, litter and debris, including shopping carts, yard waste, and so on. At the same time, urban creeks and lakes can provide important aesthetic amenities and, in some cases, recreational fisheries and nursery areas for aquatic life. The fisheries in urban creeks can range from a sustainable trout fishery to carp- or minnow-dominated waters. Some urban lakes provide good warmwater sport fisheries for bass, bluegill, and so on.

The Center for Watershed Protection report entitled, "Impacts of Impervious Cover on Aquatic Systems" (1), contains information on some aspects of the impact of urban stormwater runoff on water quality. This report is an expansion/update of earlier work by Schueler (2) on the impact of urbanization (paving) of an area on the waterbodies receiving the runoff from the area. Burton and Pitt (3) developed a *Stormwater Effects Handbook*, which provides background information on the water quality problems associated with stormwater runoff from urban and, to a lesser extent, rural areas. They discuss impacts on receiving water uses and sources of stormwater pollutants. Most of this over-900-page handbook is devoted to a discussion of approaches for

assessing the characteristics of stormwater runoff and its impacts on receiving water quality. Another source of information on the impact of urban stormwater runoff-associated chemical constituents on water quality is the *Stormwater Runoff Water Quality Science/Engineering Newsletter* available from <http://www.gfredlee.com>.

REGULATING URBAN STORMWATER RUNOFF WATER QUALITY IMPACTS

The U.S. EPA (4) promulgated the national regulations that require cities with populations greater than 100,000 to develop stormwater runoff water pollution control programs to control pollution to the maximum extent practicable (MEP) using best management practices (BMPs). These requirements are also applicable to highway stormwater runoff. The EPA did not define MEP or the BMPs that are to be used. Urban stormwater runoff pollution is to be regulated by National Pollution Discharge Elimination System (NPDES) permits that ultimately require compliance with water quality standards. Although NPDES permits for domestic and industrial wastewater discharges are to prevent violations of water quality standards at the point of discharge or at the edge of a mixing zone if provided for, the NPDES permits issued to municipalities for controlling urban stormwater pollution, while requiring ultimately compliance with water quality standards, did not specify a date by which compliance with water quality standards is to be achieved. The current regulations require the use of a BMP ratcheting-down process to ultimately, but as of yet at an undefined date, control violations of water quality standards in the runoff. Justification for this difference in approach for regulating urban stormwater runoff, compared with urban wastewater discharges, develops from the significantly different characteristics of urban stormwater runoff.

ASCE (5) and CASQA (6) list a variety of "BMPs" that are advocated for the "treatment" of urban stormwater runoff to control water pollution, including ponds that allow settling of some chemical constituents, grassy areas that allow settling and removal of some chemicals, and infiltration basins that allow infiltration of the stormwater into groundwaters. These BMPs were largely based on hydraulic factors without evaluation of their effectiveness in treating stormwater runoff to achieve water quality standards. To the contrary, it has been found that ponds (detention basins), grassy swales, and so on cannot be considered adequate for treating urban stormwater runoff to achieve compliance with water quality standards.

The cost of retrofitting conventional BMPs to developed urban areas has been estimated to be on the order of \$1 to \$3 per person per day for the population served by the storm sewer collection system. These costs are primarily for the acquisition of land and effectively restrict implementation of these BMPs to new developments where their cost can be incorporated into the cost of the development. Although conventional BMPs are being installed especially in new developments, the most popular conventional BMPs will not be adequate to treat urban or highway stormwater runoff to achieve compliance with

water quality standards. To achieve compliance with water quality standards for urban area and highway stormwater runoff, it will be necessary to construct, operate, and maintain treatment works of the type used in advanced wastewater treatment. The costs of such retrofitted treatment works in developed areas is projected to be on the order of \$5 to \$10 per person per day. These very high costs will require that a different approach be developed for regulating urban area stormwater runoff.

EVALUATION OF WATER QUALITY IMPACTS OF STORMWATER RUNOFF

Although there has been considerable work on the chemical characteristics of urban stormwater runoff, little work has been devoted to evaluating the water quality/beneficial use impacts of this runoff. Especially in light of the tremendous costs associated with providing for treatment/control of stormwater runoff, it is important to properly assess whether a chemical constituent derived from stormwater runoff that is present in an urban stream or lake is in a chemical form that is toxic or bioavailable, i.e., can cause pollution. Failure to make this evaluation can lead to expenditure of large amounts of public funds for the development and installation of so-called best managements practices that effect little or no improvement of the beneficial uses of an urban stream or lake or other waterbody receiving the stormwater runoff.

The quality of water in urban creeks, at times, is dominated by urban stormwater runoff-associated constituents. In the late 1970s/early 1980s, the U.S. EPA conducted a Nationwide Urban Runoff Program (NURP) in 28 communities across the United States. The NURP studies provided information on concentrations and loads of a variety of potential pollutants in urban stormwater runoff. Pitt and Field (7) summarized the results of the NURP studies, as did WEF/ASCE (8). Although the U.S. EPA NURP studies provided data on the concentrations and loads of a variety of potential pollutants in urban stormwater runoff, they failed to address true water quality issues, i.e., the impacts of the potential pollutants on the beneficial uses of the receiving waters for the runoff (9).

Heavy Metals

In the fall of 1998, the California Storm Water Quality Task Force conducted a review of constituents that are present in urban area and highway stormwater runoff in sufficient concentrations to cause violations of U.S. EPA water quality criteria and/or California Toxics Rule standards, which were developed by the U.S. EPA (10) for the state. Copper, lead, and zinc were found in almost all urban street and highway stormwater runoff in concentrations that would violate U.S. EPA worst-case-based water quality criteria and state standards based on those criteria. Sometimes cadmium and mercury were also present above those criteria/standards. These findings indicate that there is a potential for certain heavy metals in urban stormwater runoff to be toxic to aquatic life in urban creeks.

Lee and Taylor (11,12) presented the results of a study of heavy metal concentrations and aquatic life toxicity in ten different Upper Newport Bay (Orange County, California) watersheds during 1999–2000. Several watersheds had predominantly urban land use. Lee and Taylor found several heavy metals, including copper, zinc, and lead, in concentrations above water quality criteria/standards. Through toxicity identification evaluation (TIE) studies, Lee and Taylor (12) found, as have others, that heavy metals in urban residential area and highway stormwater runoff are in nontoxic forms. However, this is not necessarily the case for heavy metals in industrial stormwater runoff. Several examples exist in which heavy metals such as zinc from galvanized roofs or copper from copper roofs can be present in industrial stormwater runoff in sufficient concentrations and available forms to be toxic to aquatic life.

Aquatic Life Toxicity

Several constituents are normally present in urban-area stormwater runoff that could cause aquatic life toxicity. The constituents of greatest concern are the heavy metals, including copper, zinc, lead, and occasionally cadmium. Toxicity measurements of urban stormwater runoff from several areas (12–14) have shown that although runoff from urban residential and commercial areas may be toxic to *Ceriodaphnia* (a U.S. EPA standard freshwater zooplankton test organism), that toxicity has not been because of heavy metals. Toxicity identification evaluations have shown that the toxicity measured was caused by the organophosphorus (OP) pesticides diazinon and/or chlorpyrifos. Although the OP pesticides are of concern because of their toxicity to a few types of zooplankton, they are not toxic to fish or algae at the concentrations typically found in urban runoff.

Pesticide-Caused Toxicity

Lee et al. (13,14) reviewed the topic of OP-pesticide-caused toxicity. Diazinon and chlorpyrifos have been, or will soon be, phased out of urban use by the U.S. EPA because of their potential toxicity to children. Chlorpyrifos can no longer be sold for use as a pesticide in urban areas. The U.S. EPA and the registrants have agreed that it will no longer be legal to sell diazinon for urban use after December 2004. These OP pesticides are being replaced by others, especially the pyrethroid pesticides, in urban areas. However, the replacement pesticides have not been evaluated by the U.S. EPA Office of Pesticide Programs for their potential to cause aquatic life toxicity in stormwater runoff from their point of application. Several pesticides are more toxic to fish and zooplankton than the OP pesticides. Further, many of the pyrethroid pesticides tend to sorb strongly to soil particles and, therefore, will be transported in particulate form and accumulate in sediments. Weston (15) and Weston et al. (16) have reported finding that some sediment-sorbed pyrethroid-based pesticides are bioavailable to some benthic organisms. It is unclear whether this bioavailability leads to toxicity. It could, however, cause toxicity in urban streams and lakes and their sediments

that is adverse to aquatic life-related beneficial uses of the waterbody.

Dissolved Oxygen

Stormwater runoff events can cause significant dissolved oxygen (DO) depletion in urban streams and other nearby waterbodies. DO measurements made by the DeltaKeeper (17) in waterbodies just before, during, and after a runoff event showed that the DO before the event was adequate for maintenance of aquatic life, i.e., above about 5 mg/L. However, shortly after the event began, the DO in some of the waterbodies dropped to less than 1 mg/L and stayed depressed for several days. A stormwater runoff event in November 2002, and another in August 2003, which were the first major runoff events of the summer/fall, led to large fish kills in half a dozen or so of those waterbodies.

Nutrients

Urban stormwater runoff contains elevated concentrations of various nutrients (nitrogen and phosphorus compounds) that can lead to excessive fertilization of urban creeks, lakes, and downstream waterbodies. Kluesener and Lee (18) and Rast and Lee (19) determined the nutrient loads associated with urban stormwater runoff. In addition to being derived from stormwater runoff, nutrients, especially nitrate, can also be present in groundwater flow to urban creeks and lakes, which can be an important source of nitrate.

Cowen and Lee (20) reported that part of the algal available P in urban stormwater runoff was derived from the leaching of tree leaves and flowers. Cowen and Lee (21) conducted studies of the algal available phosphorus in urban stormwater runoff in several urban areas. Lee et al. (22) summarized the results of those studies and those of others on algal available P in urban and agricultural runoff. In general, it has been found that the algal available P in stormwater runoff from urban and agricultural areas is equal to the sum of the soluble orthophosphate plus about 20% of the particulate phosphorus. Therefore, about 80% of the particulate phosphorus (which can be most of the phosphorus load in such runoff) does not support algal growth.

pH

There can be sufficient primary production in urban creeks and lakes to cause significant diel (over a 24-hr day) changes in pH and dissolved oxygen, which is especially true for those urban streams that have only limited areas where extensive canopy from trees along the bank shades the water. The U.S. EPA (23) Gold Book water quality criterion limits the pH of waters to 9. It is not unusual for the pH of waterbodies to exceed that value in the late afternoon, at the height of photosynthetic activity, and to be several units lower in early morning.

Ammonia

It is possible for the ammonia concentrations in urban creeks to be sufficiently high to violate ammonia water

quality criteria based on potential toxicity to aquatic life in an urban stream or lake, which is especially likely if a significant storm sewer discharge contains storm sewer-accumulated sludge/sediments scoured during a runoff event, or scour of stream sediments, which would tend to have high ammonia concentrations. The impact of that ammonia would be exacerbated in those urban streams and lakes that are highly productive as they would tend to have an elevated pH in mid-afternoon because of photosynthetic activity.

Sanitary Quality

Especially during dry weather flow, urban stormwater runoff and, in some situations, drainage ways such as creeks in urban areas, often have greatly elevated concentrations of total coliforms, fecal coliforms, and *E. coli*. The U.S. EPA (24) announced that it was going to require that states adopt a revised contact-recreation criterion for fresh water based on the measurement of *E. coli*. *E. coli* has become the standard recommended organism for assessing the sanitary quality of a freshwater with respect to contact recreation. It is also a useful indicator of potential pathogens in domestic water supplies. Enterococci have become the standard fecal indicator organism for marine waters. The U.S. EPA (4) announced that it was implementing its 1986 criteria for those bacteria in states bordering Great Lakes and in ocean waters that had not adopted those criteria by April 2004. In 2005, the U.S. EPA will develop revised contact recreation-based water quality criteria for the inland waters of the United States.

In many communities, the design of the sanitary sewerage (collection) system is such that discharges of raw sewage to urban waterways can be associated with pump station power failure, blockage of the sewer, and other factors. Further, sanitary sewerage systems are sometimes poorly maintained, with the result that there can be discharges of raw sewage to nearby watercourses on an ongoing basis through leaks in the sewerage system. In addition, animals, including birds, can contribute significant amounts of fecal coliforms and *E. coli* to stormwater runoff, which, in turn, can cause urban creeks to have poor sanitary quality.

With increased emphasis on managing the water quality impacts of urban stormwater runoff in some parts of the country, such as Southern California (especially in the Santa Monica Bay watershed, because of the adverse impacts on sanitary quality of Santa Monica Bay beaches), efforts are being made to control *E. coli* and other pathogen indicators in stormwater runoff, as well as in separate storm sewers during dry weather flow. Ultimately, through comprehensive studies that are now being developed in the Los Angeles Basin and elsewhere, information will be gained on the specific sources of *E. coli* and the potential for their control. Information on the current understanding and control of the sanitary quality of urban stormwater runoff is available in the proceedings of the US EPA 2004 national Beaches conference: <http://www.epa.gov/beaches/>.

Total Organic Carbon (TOC)

Based on U.S. EPA regulations, domestic water supplies that have a total organic carbon (TOC) concentration above about 2 mg/L may be required to treat the water to remove the total organic carbon to that level, in order to reduce the potential for formation of trihalomethanes (THMs) and other disinfection byproducts during the disinfection of the water supply. This situation raises the question as to whether urban stormwater runoff could be a significant contributor of TOC to urban creeks and ultimately to downstream waterbodies that are used for domestic water supply purposes. Site-specific investigations need to be conducted to evaluate this situation for a particular waterbody.

Excessive Bioaccumulation of Hazardous Chemicals in Edible Aquatic Organisms

Fish and other edible aquatic organisms taken from some urban streams have been found to contain excessive concentrations of legacy pesticides such as DDT, dieldrin, and chlordane, derived from their former use in urban areas as well as from current runoff from urban areas that had been agricultural. In addition, fish and other aquatic life in urban streams can contain excessive concentrations of PCBs and dioxins/furans. As discussed by Lee and Jones-Lee (25), dioxins are known to be present in stormwater runoff from urban areas and highways and can, therefore, be present in urban streams and lakes, especially in the sediments. PCBs are sometimes found in urban stream fish because of spills of electrical transformer PCBs that have occurred in the urban stream watershed or illegal discharges of PCBs from industrial sources to the storm sewer system.

An example of this type of situation occurred in the Smith Canal in the city of Stockton, California. Some edible fish taken from that canal in 1998 contained concentrations of PCBs at levels that are considered hazardous for consumption because of the increased risk of cancer. Lee et al. (26) conducted a study on Smith Canal sediments to determine the total concentrations of PCBs in the sediments and their bioavailability using the U.S. EPA standard sediment bioavailability test procedure with *Lumbriculus variegatus*. It was found that although the sediments had high TOC, which would tend to make the PCBs less bioavailable, there still was significant uptake of the PCBs from the sediments by *Lumbriculus*, which indicates that those organisms would be a food-web source of the excessive PCBs that are found in higher trophic-level edible fish taken from parts of the Smith Canal.

Some measurements of mercury in urban stormwater runoff have shown that the concentrations are sufficient to potentially lead to excessive bioaccumulation of mercury in edible fish tissue. In urban streams or lakes where bioaccumulation of mercury is a potential concern, fish should be examined to determine if they have excessive bioaccumulation of mercury. Lee and Jones-Lee (25) and Lee (27) have provided guidance on approaches that should be followed to evaluate excessive mercury bioaccumulation by examination of edible fish tissue.

PAHs, Oil and Grease, and Unrecognized Hazardous/Deleterious Organic Chemicals

Numerous organic compounds are not pesticides or organochlorine bioaccumulatable chemicals but are of potential concern in urban stormwater runoff. These compounds include oil and grease, PAHs, and others included in the group of "total organic carbon." Within the oil and grease and TOC fractions in urban stormwater runoff can be thousands of unregulated organic chemicals that pose a threat of toxicity to aquatic life and/or bioaccumulate in edible aquatic life where they pose a threat to higher trophic-level organisms, including humans. Many chemicals have been in use and entering the environment for many years but have not been regulated. For example, Silva (28) of the Santa Clara Valley Water District, California has reported that sufficient perchlorate leaches from a flare used at a highway accident to contaminate 726,000 gallons of drinking water with perchlorate above the California Department of Health Services action level of 4 µg/L.

Daughton (29) indicated that although there are more than 22 million organic and inorganic substances, with nearly 6 million commercially available, fewer than 200 are addressed by the current water quality regulations. He noted special concern that in general pharmaceuticals and personal care products (PPCPs) are not regulated but can pose significant water quality concerns. Daughton stated, "Regulated pollutants compose but a very small piece of the universe of chemical stressors to which organisms can be exposed on a continual basis." Additional information on PPCPs is available at <http://www.epa.gov/nerlesd1/chemistry/pharma/index.htm>.

Suspended Sediment/Turbidity

If an urban creek watershed contains areas of new construction and/or if the urban creek watershed and the creek have soils that readily erode, there can be significant increases in suspended solids/turbidity in the creek during runoff events. The increased turbidity makes the water turbid (muddy), which can affect aquatic life habitat.

Trash

Urban creeks are notorious for accumulating materials that people discard, including grocery carts, tires, paper, Christmas trees and shrubbery, and lawn trimmings. Although some of these items can inhibit flow and thus lead to flooding, some of this material also provides habitat for aquatic organisms in the creek. The primary adverse impact of trash is on the aesthetic quality of the waterbody. Some creeks receive large amounts of trash, which is evidenced by the "creek days" that environmental/public groups conduct, when debris of various types is removed from the creek. With increased emphasis being placed on controlling trash in stormwater runoff in the Los Angeles area pursuant to a TMDL issued to control trash in urban stormwater runoff (30), there could be a reduction in the total amount of trash that is dumped into Los Angeles area urban creeks.

Aquatic Life Habitat

As part of its Water Quality Criteria and Standards Plan (24), the U.S. EPA specifically delineated urban stormwater runoff as a cause of deteriorated aquatic life habitat. The habitat degradation is a result of a variety of factors including channelization and increased urban stream flow caused by paved development in the watershed.

The CWP (1) report contains an extensive discussion of the impact of urbanization with the associated increase in impervious cover (e.g., paving) in urban stream watersheds, on the hydrological and morphological characteristics of urban streams. It reported that when the percentage of impervious cover in an urban stream's watershed exceeds about 10%, the stream's characteristics are typically impacted. When the impervious cover exceeds about 25%, severe impacts on the waterbody's characteristics tend to occur.

As part of the implementation of its Water Quality Criteria and Standards Plan, the U.S. EPA plans to pursue the use of bioassessment methodology to determine the degree of degradation caused by urban stormwater runoff that would need to be corrected to develop desirable aquatic life habitat in urban streams and other waterbodies that receive urban stormwater runoff (24). Thus far the U.S. EPA and state water pollution control agencies seem to have made little progress toward achieving this goal. Information on the U.S. EPA's current program in this area is presented at <http://www.epa.gov/ebtpages/watwaterbioassessment.html>.

OVERALL

It has become evident that a need exists for comprehensive water quality monitoring/evaluation programs to determine, for representative locations, the real, significant water quality-use-impairments that are occurring in urban lakes and streams (and, for that matter, downstream waters) receiving urban area and highway stormwater runoff. This monitoring/evaluation program should include defining the specific sources of the constituents that lead to the water quality/use impairments. Once the water quality problems have been defined and the sources of the responsible pollutants have been identified, then a reliable evaluation can be made of the management practices that can be implemented to control the pollution of urban streams and lakes by urban area stormwater runoff-associated constituents. In general, because of the high cost of treatment, it is likely that the management practices will focus on source control, as opposed to treatment of the stormwater runoff.

The U.S. EPA's announced "Strategy for Water Quality Standards and Criteria" (31) includes development of wet weather water quality standards. These standards would more appropriately consider how chemical constituents in stormwater runoff impact the beneficial uses of receiving waters. They would likely include a weight-of-evidence evaluation of the relationship between the concentrations of toxic/available forms of constituents in stormwater

runoff and their impacts on aquatic-life-related resources in the waterbodies receiving the runoff. Lee and Jones-Lee (32) have reviewed this approach for managing urban area stormwater runoff water quality (33).

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RIVERS AND STREAMS: ONE-WAY FLOW SYSTEM

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Broadly speaking, rivers and streams are inscribed in the structural relief lines of the lowest local altitudes, the thalwegs. But they do not necessarily strictly follow these lines when they are considered on a very precise scale, because their exact drawing is constrained by local particularities such as the substratum, the vegetation, and anthropic artifacts. Inversely, there is not necessarily a river or a stream in a thalweg because the permanent flow and the landscape marking depend on many hydrologic and geomorphological factors.

Because of this particular position of rivers and streams, they have a major role in hillslope drainage, whatever the kinds of waterways through the hillslopes are. Then, any river or stream receives water from the sides and from upward, and makes it flow downward through a slope-oriented sense. Thus, a point of a stream or a river has a topological and functional relationship with all of its up and downward points. This relationship leads to defining the so-called 'one-way flow system' and makes structured objects emerge on a larger scale. Rivers

and streams meet and are organized in a hierarchical branched network.

A drainage network is made up of particular points: upstream extremities called sources, junctions called confluences, and a unique downstream extremity called the outlet—and the identification of this outlet is directly related to the definition of a watershed. The channel joining two such points is a link (1–3). To characterize the position of any link within the system, one can identify its magnitude (4), which is the number of sources it drains. One can also attribute an order to it, according to one of the proposed ordering schemes (1,5). Particularly, according to Strahler's scheme, the notion of (network)-stream is defined as a set of successive links of the same order, from up to downstream (6).

From this identification of individual and collective characteristics of river and stream systems, many works have been performed to describe, explain, quantify, and simulate the organizational schemes and rules of the constituents and furthermore, to base the simulation of water flows on these schemes and rules.

Planar observation of individual channels led to observing different patterns and shapes, mostly based on sinuosity, braiding, and anabranching. The cross section of a given site in a channel can present very different shapes and sizes. Moreover, planar observation of drainage networks also points out different patterns. Such individual and network patterns result from complex interactions between the substratum, the vegetation, the anthropic constraints and influences, and the water fluxes. Moreover, these complex interactions are scale-dependent through a wide range of spatial, temporal, and frequency scales. Nevertheless, these patterns can be described by typologies and by quantifying some relevant geometric criteria (7,8). Among them are the sinuosity characteristics such as the meandering radius and the wavelength (9); the bankfull width, depth, and slope; and the drainage density (1,10).

Within the range of scale-dependencies, the relationship with the corresponding watershed is a major issue. First, even if the development of a drainage network follows up and down dynamics (1,6,11), the initiation of a channel has been related to planar geometric characteristics of the upward territory (1,6), called the source area. Furthermore this dependence can be linked to hydrologic processes and to slopes (12,13). Thus, the drainage density appears to depend partly on geometric and partly on hydrologic considerations (1,13,14), which generates great geographic and temporal variability (3,15).

Second, the local geometric and hydraulic characteristics of a channel are related to characteristics of the watershed at whose outlet it is located (8,16). The first such relation, found by Hack (17), is the expression of the length of the longest waterway as a power law of the area of the watershed. Then, other power law relationships have been identified and studied for a range of regimes and substratum between the discharge of a given frequency of occurrence and the area and between stream channel geometric parameters (depth, width, slope, and velocity) and the discharge.

The channel slope at any point is thus related to the upstream conditions. Along a waterway from a source to the outlet, the upstream characteristics vary, particularly the drainage area, and the channel slope presents a covariation. The longitudinal profile of a waterway demonstrates this covariation that generally presents a typical decreasing shape (18), whose characteristics and variability depend mostly on geology and climate and sometimes, on anthropic effects.

These interpretations of local characteristics within systemic trends are elements of an organization of the drainage network. Other watershed-scale elements tend to show that there is a kind of organization of the hydrogeomorphological one-way flow system. These elements come from the analysis of quantitative characteristics which are identified from geometric, topological, and hydrologic considerations.

"The pioneer work" in this domain is the identification of Horton laws of network planar composition, first proposed with Horton's ordering scheme (1) and then verified with Strahler's (6,19) by the "network definition" of a stream. These laws show that the number of streams of order i , the mean length of streams of order i , the mean total area contributing to streams of order i , and the mean slope of streams of order i broadly satisfy geometric series, according to characteristic ratios, respectively, the bifurcation, the length, the area, and the slope ratios. These laws were recognized as geometric-scaling relationships and were integrated in the fractal analysis of the river network organization (20,21), which was then applied to a wide range of parameters and functions (16,22). The width function, particularly, has both geomorphological and hydrologic significance and is deeply studied (7,16).

All these observations and quantifications of the one-way flow system of rivers and streams are characteristic of a given state of a system and can furthermore help follow the evolution of the system. The channels themselves and their relationships with upstream channels and hillslopes, and the network structure evolve according to natural trends and eventually to anthropic constraints. It appears then that the characteristic observations and quantifications tend to particular values or shapes which show a natural underlying tendency of energy optimality or of self-organized criticality (16,23).

Furthermore, on the basis of these observations and quantifications, many modelings can be proposed. First, organization models try to produce realistic networks, based on deterministic and/or statistical reasoning using topological, percolation, optimality, or self-organization rules (16). Second, hydraulic and hydrologic models can base the quantitative description of water fluxes on such actual and theoretical geomorphological considerations (7,16).

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STREAMFLOW

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A knowledge of hydrology is fundamental to decision-making processes where water is a component of the engineering system. Streamflow is an important

component of the hydrologic cycle. The primary input in a hydrologic cycle is precipitation (rainfall or snowfall). Many forms of rainfall losses occur on Earth's surface, e.g., interception by surface vegetation, depression storage, and infiltration to soil. Rainfall excess is the net rainfall after subtracting rainfall losses from gross rainfall and turns into surface runoff. Surface runoff can eventually gather into natural streams or rivers, and flows in streams/rivers are referred to as streamflow. Streamflow typically quantifies as volumetric discharge in cubic meter or feet per second (cms or cfs). Streamflow is not directly recorded, even though streamflow is one of the most important parameters in hydrologic studies. Typically, the water level at a specific stream cross section is recorded and streamflow is deduced by means of a rating curve. The rating curve is developed by using a set of discharges and gauge heights measured over a period of months or years. Discharges used for rating curve development can be determined by numerical integration after flow velocities are measured at several locations along the cross section and where velocity changes. The principal sources of streamflow data for the United States are the U.S. Geological Survey, U.S. Natural Resources Conservation Service (NRCS), U.S. Forest Service, and U.S. Agricultural Research Service.

Streamflow, at a given location on a stream, is usually represented by a hydrograph. A streamflow hydrograph is a graphical representation of instantaneous discharge at a given location with respect to time. Two types of hydrographs are particularly important: the annual hydrograph and the storm hydrograph. The annual hydrograph is a plot of streamflow versus time over a year, and it shows long-term water balance of precipitation, evaporation, and streamflow in a watershed. From the annual hydrograph, a stream may be classified as a perennial river, ephemeral river, or snow-fed river. A perennial river typical under a humid climate has continuous flow conditions year round and even during no-rainfall periods. An ephemeral river in an arid climate has long periods when the river is dry (no flow condition). A snow-fed river has the bulk of the runoff yield occurring in the spring and early summer from snowmelt, and variation of discharge is smaller than that for other types of rivers. Annual hydrographs typically contain some spikes caused by rain storms. A storm hydrograph is a streamflow hydrograph during and after a storm.

An example storm hydrograph with typical timing parameters and rainfall hyetograph is shown in Fig. 1. A hydrograph is the response of a watershed under certain rainfall input, and it is affected by the watershed's physical, geological, vegetation, and climatic features. The integration of area under a hydrograph between any two points in time gives the total volume of runoff passing the stream cross section during the time interval. A typical hydrograph consists of rising limb, crest segment, receding limb, and base flow. The rising limb depends on duration and intensity of rainfall, antecedent moisture condition, and drainage characteristics of basin. The crest segment contains a peak that represents the highest concentration of runoff that usually occurs soon after the rainfall has ended. The receding limb represents the withdrawal of the

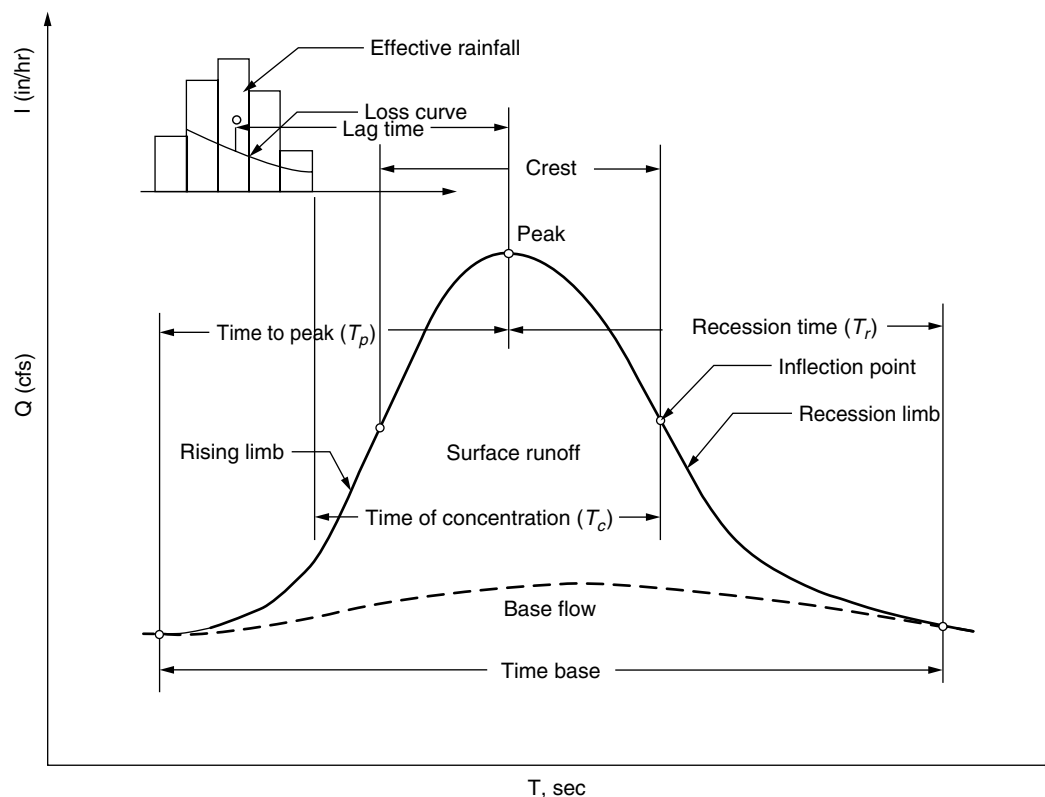


Figure 1. Schematic streamflow hydrograph including typical timing parameters and a rainfall hyetograph.

water from the storage after surface inflow to the channel has ceased. The base flow is that part of runoff that exists even before the occurrence of rainfall (e.g., in the perennial river during the no rainfall period). The total streamflow hydrograph has two components: direct runoff hydrograph (DRH) and base flow. The direct runoff hydrograph is the transformation of effective rainfall passing through a watershed. Base flow is the part of precipitation that percolates downward until it reaches ground water table and eventually discharges into the stream.

Study of a streamflow hydrograph is important for water management and environmental studies as it provides the peak flow rates and for which hydraulic structures can be designed. For the design problems, runoff volume alone is not adequate because the given volume of water may or may not present a flood hazard, but it depends on the time distribution of flood runoff. If the total volume is distributed over a long period of time, flood damages will be minimal, and if the volume is distributed for a short time, even though the volume is small, the damage may be hazardous. Several time parameters have been developed that reflect the timing of runoff, for example, lag time, time to peak, time of concentration, and recession time in Fig. 1. Although direct runoff begins with the commencement of effective rainfall (Fig. 1), the largest portion of runoff generally lags the rainfall because it takes time for runoff to travel from any location within the watershed to the outlet. The basin lag time could be defined as the time from the center of mass of the rainfall excess to the peak discharge rate on

the hydrograph (Fig. 1). The time to peak is the time from the beginning to the peak discharge in a simple (single peak) direct runoff hydrograph (Fig. 1). Both the lag time and the time to peak of a hydrograph correlate to the time of concentration of a watershed. The time of concentration (T_c) is the time it takes a water parcel to travel from the hydraulically most distal part of watershed to the outlet or reference point downstream (1–5). In hydrograph analysis, the time of concentration is the time difference between the end of rainfall excess and the inflection point of a hydrograph where the recession curve begins as shown in Fig. 1.

Generally, for a hydrological design, a hydrologist is required to provide peak rates of discharge, a stage height of a hydraulic structure, or a complete discharge hydrograph for a design frequency. The peak discharge of streamflow can be obtained from long-term historical records or from regression equations developed from recorded streamflow. A complete hydrograph, which is needed for design of reservoirs and detention and retention ponds, can be estimated from various rainfall-runoff models or regional dimensional or dimensionless (unit) hydrographs (4). The unit hydrograph is the hydrograph that results from unit rainfall excess distributed uniformly over the drainage area at a constant rate for an effective duration (6). For example, NRCS has a dimensionless unit hydrograph (DUH) procedure for design applications. This DUH developed by Victor Mockus (3) was derived from a large number of natural unit hydrographs from watersheds varying widely in size and geographical locations. Time to peak (T_p) is equal to the watershed

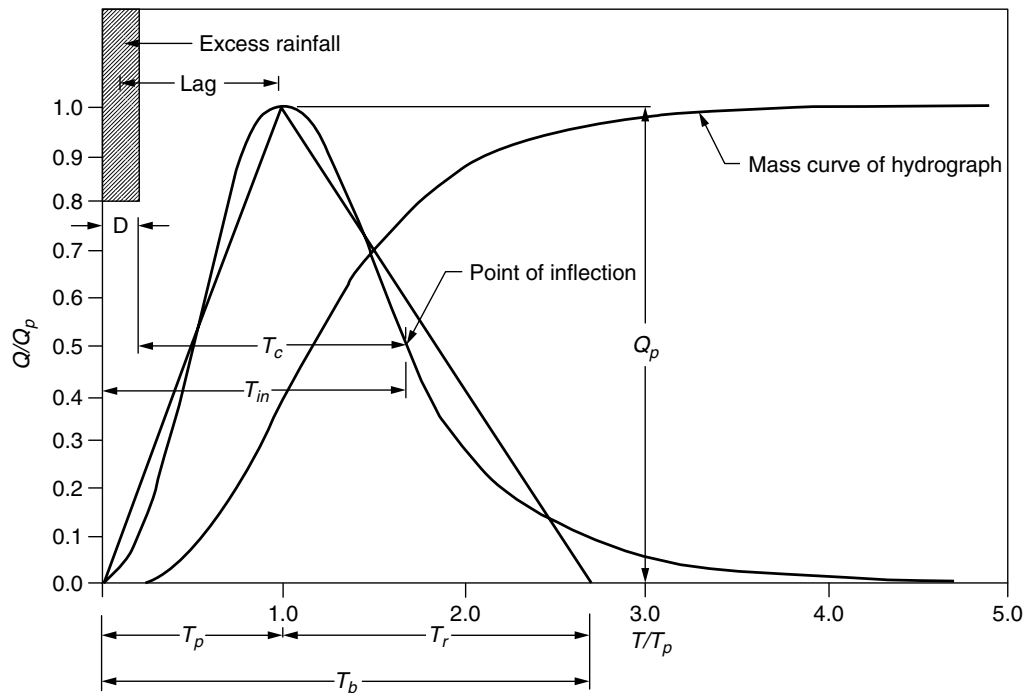


Figure 2. NRCS synthetic unit hydrographs including dimensionless hydrograph and triangular unit hydrograph.

lag time (T_L) plus half of the rainfall excess duration or the duration of unit hydrograph (D in Fig. 2).

$$T_p = T_L + D/2 \quad (1)$$

This dimensionless unit hydrograph also can be represented by an equivalent triangular hydrograph (Fig. 2). These characteristics of the NRCS unit hydrograph represent values that have been adopted for an *average* watershed. For NRCS DUH, the peak discharge (Q_p) is given as

$$Q_p = KA/T_p \quad (2)$$

where Q_p is in cfs (cubic feet per second), A (area) is in mile^2 , and T_p is in hours. K is the peak rate factor (PRF) and is considered equal to 484, assuming a triangular hydrograph with a time base being $8/3 T_p$ (Fig. 2). K is related to the internal storage characteristic of a basin and can vary considerably depending on watershed characteristics and scale (size) of a basin. For example, K can range from a value of nearly 600 for steep mountainous conditions to a value nearly to 300 in the flat coastal plains (swampy country) of the state (3). For a very flat, high-water-table watershed, the NRCS peak rate factor of 484 or even 300 likely is too large. NRCS DUH also has an empirical relationship for average lag time that is assumed to be $0.6 T_c$ (the time of concentration).

$$T_L = 0.6 T_c \quad (3)$$

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WATER QUALITY IN SUBURBAN WATERSHEDS

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The story of water quality in suburban watersheds is primarily that of nonpoint source pollution (NPS). There may be instances where point sources contribute significantly to degraded water quality in suburban areas, such as privately owned or municipal sewage treatment plants, but the most salient characteristic of suburban basins is the conversion of land from a previous forested or agricultural land cover to highly impervious surfaces such as rooftops, roads, and sidewalks.

When rain or snow strikes the earth's surface, the water may infiltrate into the soil and become part of the groundwater, or it may flow over the surface

and enter a stream system as runoff. Under natural conditions, a high proportion of precipitation soaks into the ground and does not directly enter the surface water network, although groundwater may reappear as surface water at the interface between streams and the water table. Forested watersheds can absorb huge amounts of precipitation without experiencing runoff; often one inch or more of precipitation may be absorbed into the soil (1–7). Alteration of the land surface within a watershed reduces infiltration and increases the proportion of precipitation directly entering the waterways as runoff. The increase in the percentage of impervious surface from urbanization in a stream basin is the primary threat to water quality in suburban areas. Zoppou (8) presents an up-to-date review of the numerical models currently used to simulate urban runoff.

Construction activities in land development pose a direct threat to stream quality by sedimentation from the newly cleared land surface. The removal of vegetation destabilizes the soil and allows rainfall to move large amounts of sediment rapidly into the waterways, burying the habitat for macroinvertebrates and juvenile fish and directly killing aquatic vegetation by interfering with photosynthesis (9,10) (Fig. 1). Many local governments require sediment control measures during construction to reduce damage to downstream waters, but many investigators have found that current erosion and sediment control methods are often ineffective (6,11–13).

Streams in suburban watersheds exhibit radically altered flow regimes compared to those in unaltered basins (14). The hydrographs of urbanized streams typically have higher peaks during storms and show more frequent flows above base level because precipitation that formerly soaked into the soil and joined the groundwater is now shunted into the stream network. Urban streams also have steeper rising and falling limbs of the hydrograph, as shown in Fig. 2. More frequent and rapid flooding is known as “flashiness.” These effects destabilize the streambed,



Figure 1. Suburban streams typically have entrenched, widened channels with unstable banks.

which had reached a state of dynamic equilibrium in the predevelopment condition (9,15). The result is erosion of the bed and banks and accompanying increases in sedimentation and direct destruction of habitat for aquatic flora and fauna. Other physical alterations include increased temperature and increase in sediment load. The effects of velocity and other hydraulic parameters on stream fauna are presented cogently and entertainingly by Vogel (16).

Suburban development affects surface water chemistry as well, and suburban watersheds are second only to agricultural areas in degraded water quality from chemical contributions. Nutrients such as phosphorus and nitrogen are often increased by higher population density in a watershed. The use of lawn fertilizers may allow runoff of high concentrations of both elements during storms, and home users of these chemicals often apply amounts far in excess of the capacity of the turf to absorb and use them (3). Golf courses are often sources for these nutrients as well

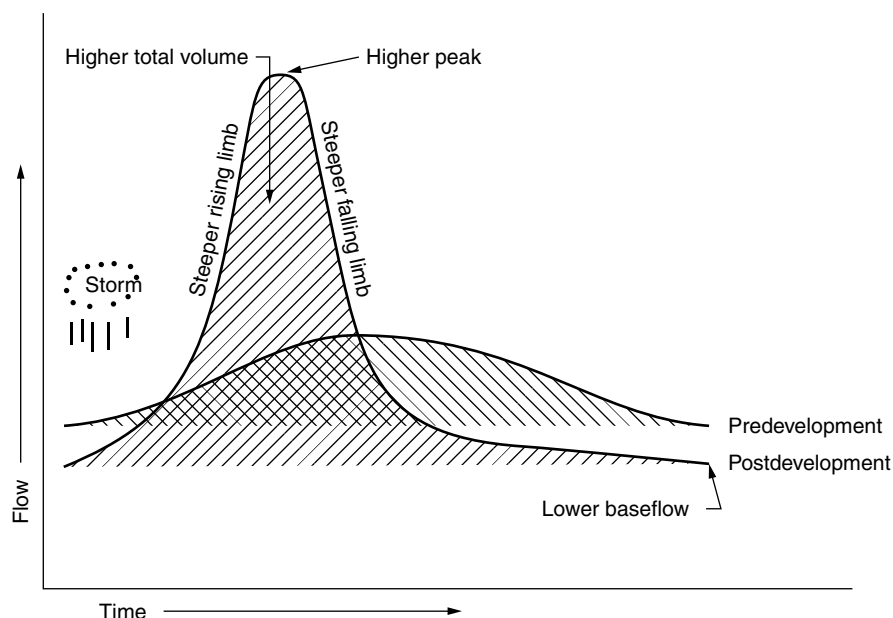


Figure 2. Increased impervious surface in suburban watersheds causes higher volume of runoff, higher peak flow, and lower base flow levels than in undeveloped watersheds.

as other pollutants in suburban areas. Nitrogen has no federally mandated maximum concentration, but such a limit has been set for phosphorus at 1 part per million by the Clean Water Act.

A number of metals are found in elevated concentrations in suburban surface waters. These include nickel, zinc, lead, mercury, cadmium, and copper. The sources of these pollutants range from dry deposition of lead and mercury to particles from automobile brakes and other systems in other cases. Metal concentrations rarely reach the acutely toxic levels found in surface waters drained from industrial sites, but suburban metals pollution nevertheless contributes to degradation of the macroinvertebrate fish, and periphyton populations. In lentic communities and higher order lotic habitats, adsorbed metals may remain permanently in bed sediments (17).

Current stormwater management practices codified in zoning ordinances and municipal facilities standards manuals concentrate almost exclusively on control of erosion and sedimentation (E&S). This emphasis can be traced to the early awareness of civil engineers and water resource managers of the loss of topsoil caused by poor agricultural and construction management practices in the first half of the twentieth century (18). The measures implemented in the period following World War II in most parts of the country are a significant improvement over the prior lack of attention to stream quality issues, but they have proven quite inadequate to preserve aquatic habitat, biotic diversity, and water quality. Engineering solutions to stormwater management problems go under the rubric of best management practices (BMPs). Erosion and sediment control policies for Fairfax County Virginia, an urban county near Washington, DC, are typical of jurisdictions that have recognized water and stream quality problems and the political will to address those problems through engineering standards. Fairfax County BMP requirements for E&S control during construction include

- Design and construction of detention ponds to limit postdevelopment runoff volume from the entire site to levels no higher than the predevelopment condition. The predevelopment and postdevelopment conditions must be demonstrated by hydrologic and hydraulic modeling, according to accepted standards.
- Design of outfall from any stormwater management structures must meet maximum velocity standards to prevent scouring and erosion of the natural stream channel receiving discharge from the constructed structures. The maximum permissible velocity must be hydraulically modeled prior to plan approval and must take into account the soil type in the receiving natural stream channel.
- Temporary E&S controls during construction such as temporary stormwater detention ponds and installation of silt fences intended to prevent sediment-laden runoff from denuded areas from entering off-site streams.
- Prohibitions or limitations on development within floodplains, wetlands, or within 100 feet of a tributary stream.

Environmental regulators in Fairfax County have sent letters to engineering firms and land development companies encouraging the use of innovative BMPs such as bioretention, dry swales, filtration structures, and extended detention ponds, which offer advantages in emulating the predevelopment surface water and groundwater hydrology, but does not require them. However, it is often perceived that innovative BMPs entail higher installation and maintenance costs and are rarely used in many localities. This condition may be expected to change as programs such as the Stage II National Pollutant Discharge Elimination System and the Total Maximum Daily Loads efforts advance to the implementation stage during the next several years. Both programs will place increasing responsibility on local regulatory agencies to decrease negative impacts on receiving waters from the full range of urban hydrology characteristics.

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SURFACE WATER POLLUTION

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INTRODUCTION

Water is a basic necessity for human survival and sustenance of civilization. The ever-increasing population, steadily rising irrigation activities, rapidly expanding industries, growing urbanization, and rising standard of living exert tremendous pressure on the available water resources (1). The temporal and spatial distribution of the quantity and quality of these resources are highly uneven. Water is used for irrigation, industry, power generation, drinking, bathing, recreation, fisheries, wildlife propagation, pollution abatement, etc. Water in appropriate quantities of specified quality is required for each use (2). Therefore, the quantity as well as quality of river flow should be monitored in river management program.

Surface water pollution is a major environmental concern all over the world (3). The simplest definition of surface water pollution is “the loss of any of the actual or potential beneficial uses of water caused by any change in its composition due to human activity.” The beneficial uses of water are varied and if water is rendered unsuitable for any of the purposes as indicated earlier, then it is polluted to a greater or lesser degree, depending on the extent of the damage caused. Pollution eventually diminishes the aesthetic quality of lakes and rivers. Even more seriously, when contaminated water destroys aquatic life

and reduces its reproductive abilities, then it eventually menaces human health.

Natural waters cannot be expected to be immediately suitable for the whole range of beneficial uses. For instance, nobody should expect to drink the water of any river without treatment. Depending on the local usage, weather, and other factors, the water, in its natural state, may be turbid or very highly colored and, hence, unattractive to the user. More importantly, river water is most unlikely to be microbiologically pure, that is, free of those minute organisms (bacteria and the like) that can cause disease in humans. In these cases, water pollution means that, because of some human activity, the water is no longer suitable for some potential use, such as drinking, even after treatment. To illustrate, if a river from which water is taken for treatment for drinking receives a discharge of chemical or possibly toxic waste, the water may be rendered completely unsuitable for eventual distribution. The local treatment works are usually designed to remove natural contaminants—coloring matter, particles causing turbidity, micro-organisms, etc—and therefore may be unable to cope with a heavy load of completely foreign matter, such as chemical waste.

SOURCES OF POLLUTION

The surface water pollution in rivers, lakes, reservoirs, coastal areas, and other surface water resources may occur mainly because of point-source pollution and nonpoint-source pollution. The term point-source pollution refers to pollutants discharged from one discrete location or point, such as an industry or municipal wastewater treatment plant, to surface water resources (Fig. 1). The effluent outfall from factories, refineries, waste treatment plants, etc. emits fluids of varying quality directly into surface waters.

The term nonpoint-source pollution refers to the discharge of pollutants that cannot be identified as coming from one discrete location or point. The pollution outflow from nonpoint source (NPS) pollution, or diffuse pollution, include contaminants that enter the water supply from soils/groundwater systems and from the atmosphere via rain water. Soils and groundwaters contain the residue of



Figure 1. Polluted river in United Kingdom.

human agricultural practices (fertilizers, pesticides, etc.) and improperly disposed of industrial wastes.

A majority of pollution instances are caused by sudden or continuing, accidental or deliberate, discharge of polluting material, which on first consideration might not seem harmful or offensive at all. Such pollution events are caused by the discharge of nontoxic organic matter waste from creameries, sewage (treated or untreated), manure slurry, food production waste, chemicals, and silage effluent to water bodies. When an uncontrolled discharge of organic material occurs, the constraining factor on the growth of the bacterial pollution is removed at a stroke. An immediate abundance of food exists and a corresponding plenitude of dissolved oxygen initially. Bacterial growth is promptly stimulated and the population increases rapidly, consuming the available oxygen as it does so.

The growth of bacteria, therefore, tends to reduce the amount of oxygen dissolved in the water. The extent of oxygen depletion that occurs depends on the rapidity with which the stream takes up oxygen from the atmosphere, i.e., its re-aeration capacity. This capacity is greatest in fast-flowing, turbulent streams and least in deep, slow-flowing rivers. In addition, the loss of oxygen may be counteracted by the photosynthesis of green plants, which produce oxygen during daylight. Where the degree of pollution is severe, these compensating factors may be insufficient to prevent the oxygen content of the water decreasing to very low levels, or, in the worst case, to anaerobic conditions, where a complete absence of free oxygen exists. In such conditions, foul-smelling compounds such as hydrogen sulphide may be formed.

Deoxygenation is the most important potential effect of organic waste discharge. However, toxic compounds, such as ammonia, may be present in such a waste, particularly where they have been stored for some time and have become septic, which can happen with farm waste, in which the presence of compounds, such as ammonia and hydrogen sulphide, probably contribute as much as the deoxygenating capacity of these wastes to the devastating effect on fish stocks in pollution incidents. When toxic pollution occurs, the effects are often direct and immediately apparent—fish are killed, the flora and fauna of the water receiving the pollution may be wiped out, different visible effects will be noticeable, and there may be noxious smells. The principal effects are those of direct poisoning by the offensive pollutants discharged.

EFFECTS OF POLLUTION ON DRINKING WATER

The surface water pollution occurring because of point and/or nonpoint sources of pollution creates a health hazard in many ways. Table 1 shows the effect of various water quality variables for drinking purposes.

TREATMENT OF POLLUTANTS

For many years, the main goal of treating municipal wastewater was simply to reduce its content of suspended solids, oxygen-demanding materials, dissolved inorganic compounds, and harmful bacteria. In recent years,

however, more stress has been placed on improving means of disposal of the solid residues from the municipal treatment processes. The basic methods of treating municipal wastewater fall into three stages: primary treatment, including grit removal, screening, grinding, and sedimentation; secondary treatment, which entails oxidation of dissolved organic matter by means of using biologically active sludge, which is then filtered off; and tertiary treatment, in which advanced biological methods of nitrogen removal and chemical and physical methods such as granular filtration and activated carbon absorption are employed. The handling and disposal of solid residues can account for 25–50% of the capital and operational costs of a treatment plant. The characteristics of industrial wastewater can differ considerably both within and among industries. The impact of industrial discharges depends not only on their collective characteristics, such as biochemical oxygen demand and the amount of suspended solids, but also on their content of specific inorganic and organic substances. Three options are available in controlling industrial wastewater. Control can take place at the point of generation in the plant; wastewater can be pretreated for discharge to municipal treatment sources; or wastewater can be treated completely at the plant and either reused or discharged directly into receiving waters. Figure 2 illustrates the treatment procedure of polluted water.

Raw sewage includes waste from sinks, toilets, and industrial processes. Treatment of the sewage is required before it can be safely buried, used, or released back into local water systems. In a treatment plant, the waste is passed through a series of screens, chambers, and chemical processes to reduce its bulk and toxicity. The three general phases of treatment are primary, secondary, and tertiary. During primary treatment, a large percentage of the suspended solids and inorganic material is removed from the sewage. The focus of secondary treatment is reducing organic material by accelerating natural biological processes. Tertiary treatment is necessary when the water will be reused; 99% of solids are removed and various chemical processes are used to ensure the water is as free from impurity as possible.

INDIAN SCENARIO OF SURFACE WATER POLLUTION

Water Stress and Water Scarcity in India

In India, because of uneven distribution of rainfall, the available per capita water resources have been varying in different river basins. As a result, despite good annual rainfall, some river basins fall in the category of water-scarce and water-stressed regions, whereas many others suffer from absolute scarcity. The water availability in any region or country is reflected by “water stress index” (4). This index is based on the minimum per capita water required for basic household needs and to maintain good health. A region whose renewable fresh water availability is below 1700 cubic meters/capita/annum is a “water stress” region, whereas the one whose availability falls below 1000 cubic meters/capita/annum experiences “water scarcity.” The current national average per capita water

Table 1. Effects of Water Quality Parameters for Drinking Purposes

S. No.	Parameter	Prescribed Limits IS:10500, 1991		Probable Effects
		Desirable	Permissible Limit	
1.	Color (hazen unit)	5	25	Makes water aesthetically undesirable
2.	Odor	Essentially free from objectionable odor		Makes water aesthetically undesirable
3.	Taste	Agreeable		Makes water aesthetically undesirable
4.	Turbidity (NTU)	5	10	High turbidity indicates contamination/pollution
5.	PH	6.5	8.5	Indicative of acidic or alkaline waters, affects taste, corrosivity, and the water supply system
6.	Hardness as CaCO ₃ (mg/l)	300	600	Affects water supply system (scaling). Excessive soap consumption, calcification of arteries. There is no conclusive proof, but it may cause urinary concretions, diseases of kidney or bladder, and stomach disorders
7.	Iron (mg/l)	0.03	1.00	Gives bitter sweet astringent taste. Causes staining of laundry and porcelain. In traces, it is essential for nutrition
8.	Chloride (mg/l)	250	1000	May be injurious to some people suffering from diseases of heart or kidneys. Taste, indigestion, corrosion, and palatability are affected
9.	Residual Chlorine (mg/l) only when water is chlorinated	0.20	—	Excessive chlorination of drinking water may cause asthma, colitis, and exzema
10.	Total Dissolved Solid (mg/l) (TDS)	500	2000	Palatability decreases and may cause gastro-intestinal irritation in humans; may have laxative effect particularly on transits and corrosion; may damage water system
11.	Calcium (Ca) (mg/l)	75	200	Causes encrustation in water supply system. Although insufficiency causes a severe type of rickets, excess causes concretions in the body such as kidney or bladder stones and irritation in urinary passages. (Essential for nervous and muscular system, cardiac functions, and in coagulation of blood)
12.	Magnesium (Mg) mg/l	30	100	Its salts are cathartics and diuretic. High conc. may have laxative effect particularly on new users. Magnesium deficiency is associated with structural and functional changes. It is essential an activator of many enzyme systems
13.	Copper (Cu) mg/l	0.05	1.50	Astringent taste but essential and beneficial element in human metabolism. Deficiency results in nutritional anemia in infants. Large amount may result in liver damage, cause central nervous system irritation, and depression. In water supply, it enhances corrosion of aluminium in particular
14.	Sulphate (SO ₄) mg/l	200	400	Causes gastro-intestinal irritation along with Mg or Na, can have a cathartic effect on users, concentration of more than 750 mg/l may have laxative effect along with Magnesium
15.	Nitrate (NO ₃) mg/l	45	100	Cause infant methaemoglobinaemia (blue babies) at very high concentration, causes gastric cancer and adversely affects central nervous system and cardiovascular system
16.	Fluoride (F) mg/l	1.0	1.50	Reduces dental carries, very high concentration may cause crippling skeletal fluorosis

Table 1. (Continued)

S. No.	Parameter	Prescribed Limits IS:10500, 1991		Probable Effects
		Desirable	Permissible Limit	
17.	Cadmium (Cd) mg/l	0.01	—	Acute toxicity may be associated with renal, arterial hypertension, itai-itai disease (a bone disease). Cadmium salt causes cramps, nausea, vomiting, and diarrhea
18.	Lead (pb) mg/l	0.05	—	Toxic in both acute and chronic exposures. Burning in the mouth, severe inflammation of the gastrointestinal tract with vomiting and diarrhea; chronic toxicity produces nausea, several abdominal pain, paralysis, mental confusion, visual disturbances, anemia, etc.
19.	Zinc (Zn) mg/l	5	15	An essential and beneficial element in human metabolism. Taste threshold for Zn occurs at about 5 mg/l, imparts astringent taste to water
20.	Chromium (Cr + 6) mg/l	0.05	—	Hexavalent state of chromium produces lung tumors, can produce cutaneous and nasal mucous membrane ulcers and dermatitis
21.	Boron (B) mg/l	1.00	5.00	Affects central nervous system, its salt may cause nausea, cramps, convulsions, coma, etc.
22.	Alkalinity mg/l CaCO ₃	200	600	Imparts distinctly unpleasant taste, may be deleterious to humans in presence of high pH, hardness, and total dissolved solids
23.	Pesticides mg/l	Absent	0.001	Imparts toxicity and accumulates in different organs of human body affecting immune and nervous systems, may be carcinogenic
24.	Phosphate (PO ₄) mg/l	No guideline		High concentration may cause vomiting and diarrhea; stimulate secondary hyperthyroidism and bone loss
25.	Sodium (Na)	No guideline		Harmful to persons suffering from cardiac, renal, and circulatory diseases
26.	Potassium (K) mg/l	No guideline		An essential nutritional element, but in excess is cathartic
27.	Silica (SiO ₂) mg/l	No guideline		—
28.	Nickel (Ni) mg/l	No guideline		Nontoxic element but may be carcinogenic in animals; can react with DNA resulting in DNA damage in animals
29.	Pathogens (a) total coliform (per 100 ml) (b) Fecal coliform (per 100 ml)	1	10	Causes water borne diseases like coliform, jaundice, typhoid, cholera, etc.; produces infections involving skin mucous membrane of eyes, ears, and throat

availability figure per annum is 2464 cubic meters, implying that India is not yet in the “water stress” range. However, this is only the national average figure. There are several parts of India that are water stressed (Fig. 3)—the regions in the Indus, Krishna, and Ganga subbasins. Regions under east-flowing rivers between Mahanadi and Pennar and west-flowing rivers of Kachchh and Kathiawar are experiencing water scarcity, whereas the regions under east-flowing rivers between Pennar and Kanyakumari are suffering with absolute water scarcity (situation where the per capita availability falls below 500 cubic meters/annum). The per capita water availability here falls as low as 411 cubic meters (5), as shown in Fig. 3. The annual rainfall received by India is unevenly

distributed across its different parts and across different times of the year. As a result despite good annual rainfall, some river basins fall in the category of water-scarce and water-stressed regions, whereas many others suffer from absolute scarcity.

Surface Water Pollution in India

River Water Pollution. In addition to water scarcity and water stress, many rivers are being used indiscriminately for disposal of municipal, industrial, and agricultural wastes, thereby polluting the river water beyond the permissible limits; Fig. 4 shows major rivers.

As can be seen from Fig. 2, the river water is unfit even for irrigation purposes at some places. Thus, it has

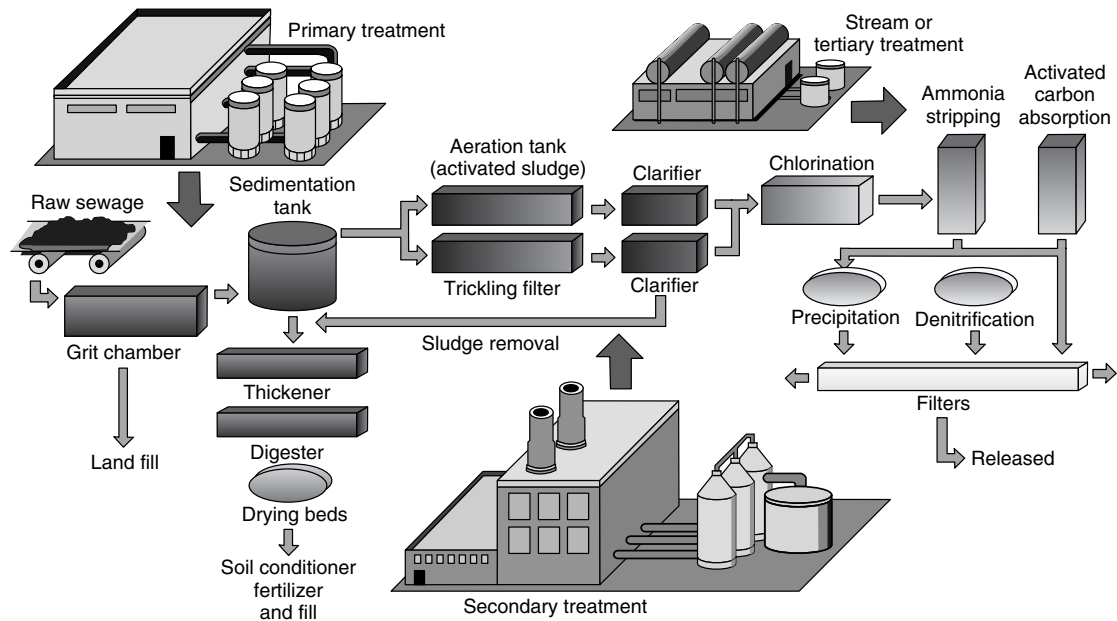


Figure 2. Wastewater treatment.

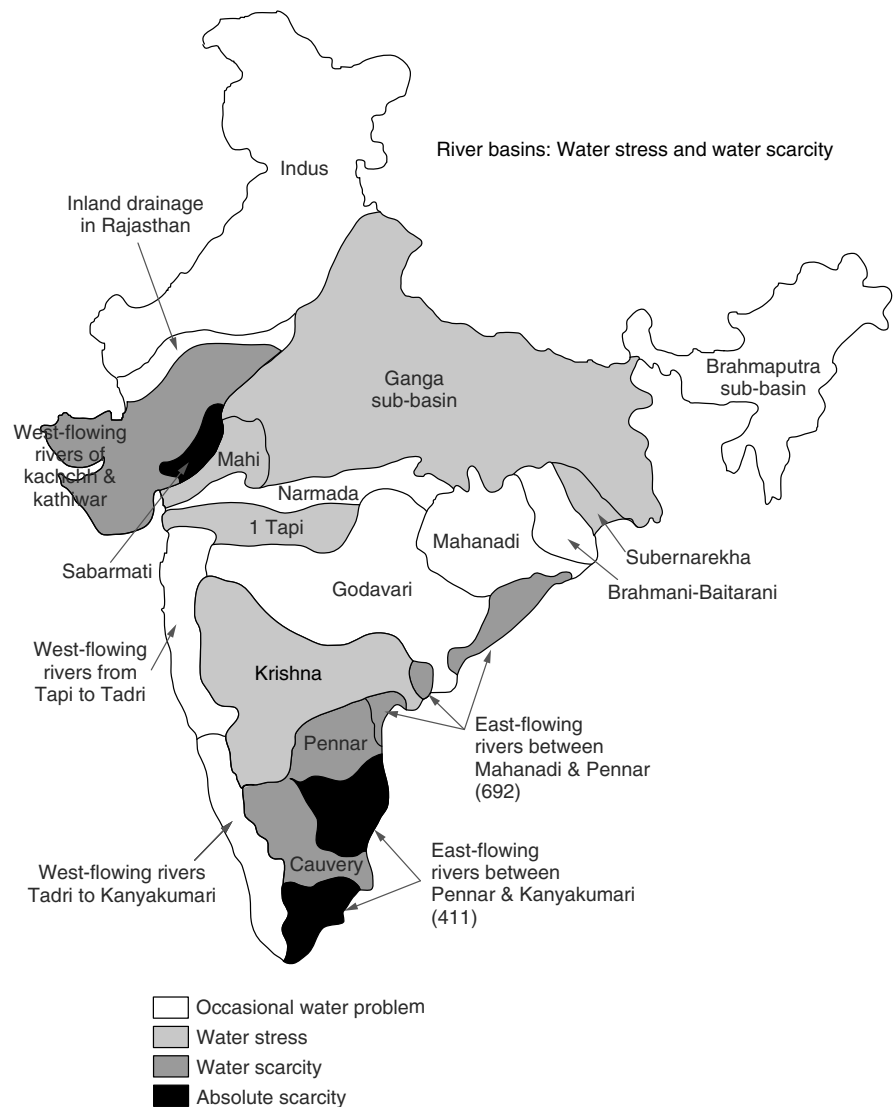


Figure 3. Water stress and water scarcity in India.

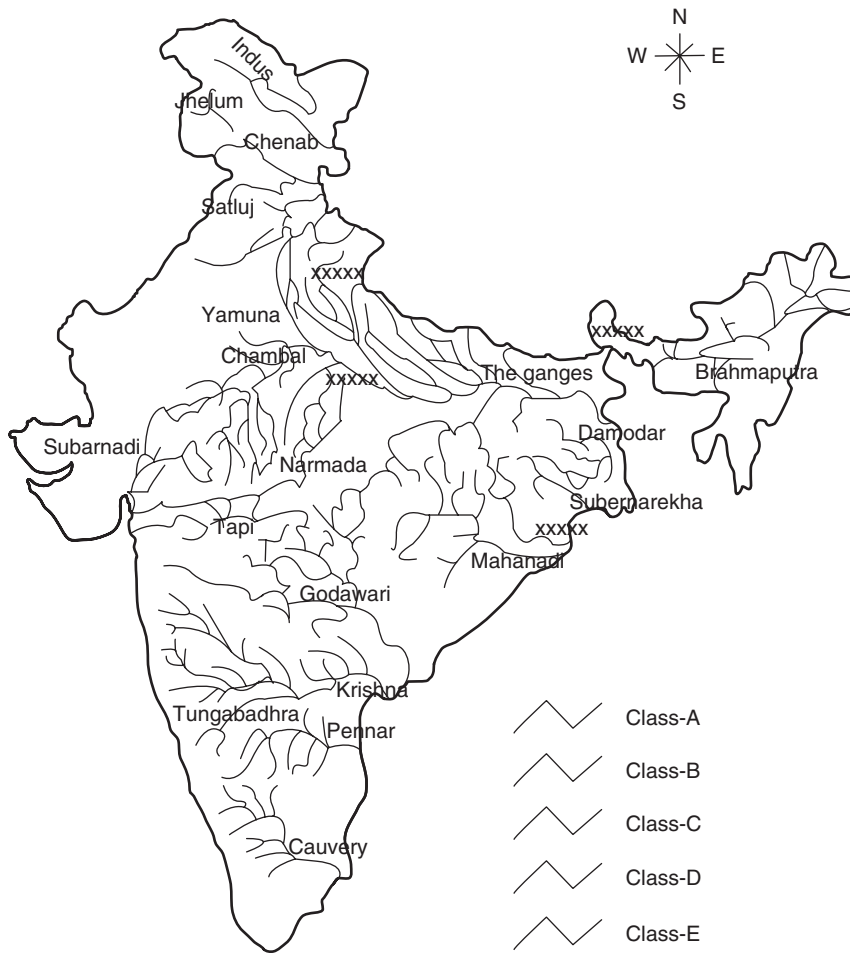


Figure 4. Generalized pattern of major rivers in India. Rivers will all be classified A to E.

Class A: Fit for drinking after proper disinfection with the addition of chlorine or bleaching powder.

Class B: Fit for bathing.

Class C: Fit for drinking only after proper treatment (screening to remove physical matters or particulate such as paper, plastic etc.

Class D: This is fit for fish and wildlife.

Class E: Suitable only for industrial cooling, irrigation, etc.

become absolutely essential to evaluate the environmental impacts of surface water resources to minimize the progressive deterioration in water quality. Water quality issues needing to be addressed with respect to rivers are discussed below.

Change in Physical Characteristics. Temperature, turbidity, and total suspended solids (TSS) in rivers can be greatly affected by human activities such as agriculture, deforestation, and the use of water for cooling. For example, the upward trend in soil erosion and the related increase in TSS in rivers can be seen in most of the mountainous regions in India (6).

Contamination by Fecal and Organic Matter. In India, fecal contamination is still the primary water quality issue in rivers, especially where human and animal wastes are not adequately collected and treated. Although this applies to both rural and urban areas, the situation is probably more critical in fast-growing cities.

The release of untreated domestic or industrial wastes high in organic matter into rivers results in a marked decline in oxygen concentration (sometimes resulting in anaerobic conditions) and an increase in ammonia and nitrogen concentrations downstream of the effluent input. The most obvious effect of the release of organic matter along the length of the river is the depletion of

oxygen downstream of the discharge as shown by the so-called “oxygen-sag curve,” which plots dissolved oxygen concentration against distance. Industrial activities that discharge large organic loads include pulp and paper production and food processing. Fecal matter affects the use of water for drinking water source or bathing water, as well as ecological health of river.

Toxic Pollutants: Organics and Heavy Metals. Organic pollutants (mostly chemicals manufactured artificially by man) are also becoming an important water quality issue. Uncontrolled discharge of industrial wastewater often causes pollution because of toxic metals. Other sources of metal pollution are leachates from urban solid waste landfills and mining waste dumps.

Rivers such as the Yamuna, which pass through large towns and cities, are often badly affected with organic pollutants. Another example is that of Damodar River, which is polluted with heavy metals developing mostly from electroplating, tanning, and metal-based industries (6).

River Eutrophication. During the 1950s and 1960s, eutrophication (nutrient enrichment leading to increased plant and algae growth) was observed mostly in lakes and reservoirs. Since the 1970s, the increasing levels of phosphates and nitrates entering rivers, particularly

Table 2. Grossly Polluted and Less Polluted Stretches of Some Major Rivers

Basin	River/ Tributary	Polluted Stretch Class	Existing Class	Desired Parameters	Critical
<i>I. Grossly Polluted Stretches</i>					
Ganga	Yamuna	(i) Delhi to confluence with Chambal	Partly D	C	DO, BOD, coliforms
		(ii) In the city limits of Delhi, Agra, and Mathura	Partly E -do-	B	-do-
	Chambal	D/s of Nagda and D/s Kota (approx 15 km for both places)	Partly D Partly E	C	BOD,DO
	Damodar	D/s of Dhanbad to Haldia	Partly D Partly E	C	BOD, Toxic
	Gomti	Lucknow to confluence with Ganga	Partly D Partly E	C	DO, BOD, coliforms
	Kali	D/s of Modinagar to confluence with Ganga	Partly D Partly E	C	-do-
	Khan	(i) In the city limits of Indore	E	B	-do-
		(ii) D/s of Indore	E	D	-do-
	Kshipra	(i) In the city limits of Ujjain	E	B	-do-
		(ii) D/s of Ujjain	E	D	-do-
	Hindon	Saharanpur to confluence with Yamuna	E	D	DO, BOD, Toxic
Godavari	Godavari	(i) D/s of Nasik to Nanded	Partly D	C	BOD
		(ii) City limits of Nasik and Nanded	Partly E -do-	B	BOD
Krishna	Krishna	Karad to Sangli	Partly D Partly E	C	BOD
Subemarekha	Subemarekha	Hathi dam to Baharagora	Partly D Partly E	C	DO, BOD, coliforms
Sabarmati	Sabarmati	Immediate upstream of Ahmedabad city up to Sabarmati Ashram	E	B	DO,BOD, coliforms
<i>II. Less Grossly Polluted Stretches</i>					
Ganga	Betwa	Between Vidisha and Mandideep and Bhopal	D	C	BOD, Total coliforms
Krishna	Krishna	Dhom dam to Narso Babri (Mah)	D	C	BOD and coliforms
		Tributary streams	D	C	-do-
		Up to Nagarjunasagar dam	D	C	-do-
		From Nagarjunasagar dam to upstream of Repella (AP)	D	C	-do-
	Bhadra	Origin to downstream of KICCL of Bhadra dam (Karnataka)	D	C	Total coliforms
	Tunga	Thirthahalli to confluence with Bhadra	C	B	Total coliforms
Cauvery	Cauvery	i) From Talakaverito 5 km of Mysore District	C	A	Total coliforms DO, BOD, coliforms BOD, Total coliforms Total coliforms
		BorderYagni (Karnataka)			
		ii) From KR Sagar Dam to Hogenekkal (Karnataka)	E	C	
		iii) From Pugalur to Grand Anicut (Tamil Nadu)	E	C	
		iv) Grand Anicut to Kumbhakonam (Tamil Nadu)	E	C	
Brahmani	Baitami	Upstream of Chandbali	D	B	BOD and coliforms
Baitami	Brahmani	Upstream of Dharmshalla	D	B	BOD and coliforms
Tapi	Tapi	From city limits of Neapanagar to the city limits of Burhanpur (MP)	E	A	DO,BOD

Source: Central Pollution Control Board, New Delhi.

in developed countries, were largely responsible for eutrophication occurring in running waters. In India, isolated reports have appeared for some river reaches, especially in plains around agriculture tracts of land.

In small rivers, eutrophication is said to promote macrophyte (large plants) development, whereas in large rivers, phytoplankton (algae) are usually more dominant than macrophyte. In such situations, the chlorophyll concentration of the water may reach extremely high values because of the fact that this pigment is present in all plants. Eutrophication can result in marked variations in dissolved oxygen and pH throughout the day. The changes in water quality caused by eutrophication can be a major cause of stress to fish because of the release, at high pH, of highly toxic gaseous ammonia and depletion of oxygen after sunshine hours.

Salinization. Industrial and mining waste pollution results in increase in specifications. Evaporation, however, increases the concentration of all ions.

Table 2 shows the water quality of grossly polluted and less polluted major rivers of India. It is found that most of the rivers lie under the category C, which is suitable only for irrigation purposes.

Lake Water Pollution. Lakes serve as traps for pollutants carried by rivers and groundwater draining the watershed. The pollutant concentration in the lake usually builds up because of evaporation of water from the lake's surface unless a natural flushing with good quality water occurs.

Eutrophication. Simply speaking, eutrophication is the biological response to excess nutrient input to a lake. The production of biomass and its death and decay results in a number of effects, which individually and collectively result in impaired water use. The most important of these effects are decreased dissolved oxygen levels, release of odorous compounds (e.g., H_2S), and siltation.

Many important lakes in India [e.g., Hussein Sagar (Hyderabad), Nainital (Uttar Pradesh) and Dal (Jammu and Kashmir)] have reportedly progressed to advanced eutrophication levels (6).

Lake Acidification. One of the major issues related to lakes in particular, and to freshwaters in general, is the progressive acidification associated with deposition of rain and particulates (wet and dry deposition) enriched in mineral acids. The problem is characteristic of lakes in specific regions of the world that satisfy two major critical conditions: the lakes must have soft water (i.e., low hardness, conductivity, and dissolved salts) and be subjected to "acid rain."

To date, lake acidification has not been reported as a problem in India.

Bioaccumulation and Biomagnification. The processes of bioaccumulation and biomagnification are extremely important in the distribution of toxic substances (discharged in waste effluents) in fresh water ecosystems. The concentration of pollutants within the organism because

of bioaccumulation and biomagnification depends on the duration of exposure of the organism to the contaminated environment and its trophic level in the food chain. Several fold increases in trace contaminant concentrations have been commonly observed in lakes and estuarine environments.

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SURFACE RUNOFF AND SUBSURFACE DRAINAGE

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Runoff is the movement of water across or through the land surface. Sometimes, the word runoff is used simply to refer to river discharge. However, there are many different types of runoff, including overland flow (surface runoff) or throughflow (subsurface runoff, sometimes also called interflow). Figure 1 illustrates the main hillslope runoff pathways for water. Precipitation can either hit the surface of the hillslope directly or be intercepted by vegetation. This intercepted water can be stored on leaves and tree trunks which shelter the ground beneath, or it can trickle down to reach the surface via stem flow. There are then two possibilities for direct precipitation or stem flow, once it reaches the hillslope, either to infiltrate into the soil or to pond up and flow over the surface as overland flow. Water that enters the soil may percolate down or move laterally

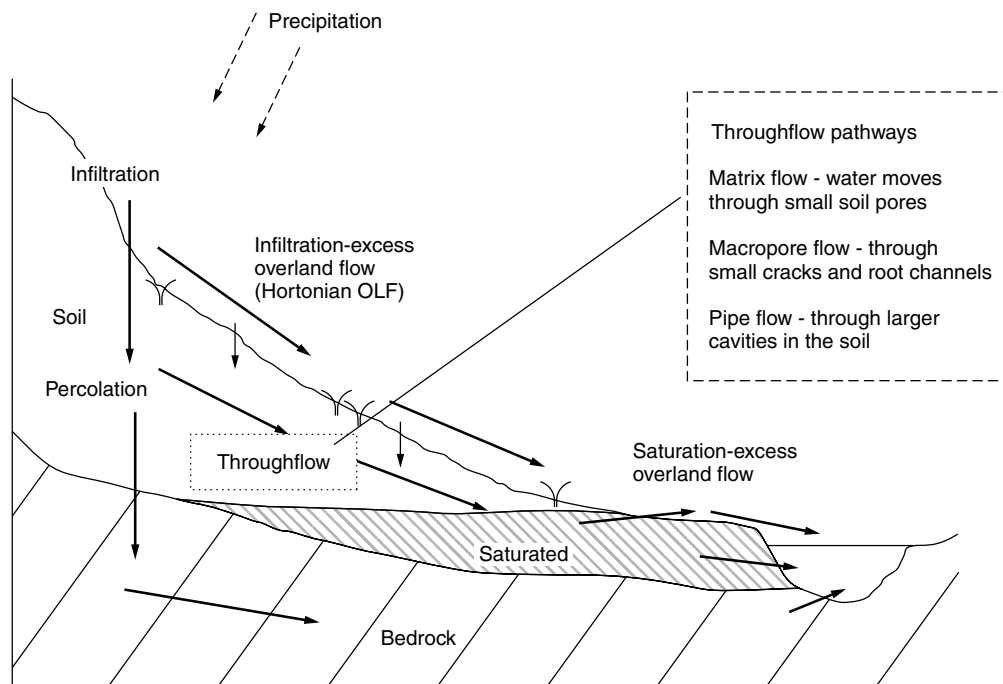


Figure 1. Main hillslope runoff pathways.

through the soil as throughflow, or it may return to the surface where the soil becomes saturated. The following sections provide details on each of the main interlinked overland flow and subsurface drainage mechanisms.

INFILTRATION-EXCESS OVERLAND FLOW

If the surface water supply is greater than the rate of infiltration into the soil, then surface storage will occur (even in urban catchments in small surface depressions). When the surface depressions are filled, they will start to overflow; this is called Hortonian overland flow (OLF) or infiltration-excess OLF. In Horton's theory, infiltration divides rainfall into two parts. One part goes via OLF to the stream channel; the other goes initially into the soil and then through groundwater flow to the stream or into groundwater storage or is lost by evapotranspiration. The dominance of Horton's (1) theory meant that research into subsurface flow mechanisms was neglected until the 1970s as it was assumed that infiltration-excess OLF was the major contributor to fast responding flow in rivers (2).

In many temperate areas of the world, infiltration-excess OLF is a rare occurrence except in urban locations. The infiltration capacity of many soils is too high to produce infiltration-excess overland flow. Infiltration-excess OLF is more likely in semiarid areas where soil surface crusts have developed and rainfall can be particularly intense. Often infiltration-excess OLF will occur only on spatially localized parts of a hillslope such as in tractor wheelings on arable land. This spatially localized infiltration-excess OLF is known as the partial contributing area concept (3). This suggests that only parts of the catchment or hillslope contribute infiltration-excess OLF, rather than the whole catchment, as Horton had originally suggested.

SATURATION-EXCESS OVERLAND FLOW

When water infiltrates into a soil, it fills up the available pore spaces. When all the pore spaces are full, the soil is saturated, and the water table is at the surface. Therefore, any extra water has difficulty entering the soil because it is saturated. Hence, OLF occurs. This type of OLF is known as saturation-excess overland flow. Saturation-excess OLF can occur at much lower rainfall intensities than required to generate infiltration-excess OLF. Saturation-excess OLF can occur even when it is not raining. This might happen, for example, at the foot of a hillslope. Water draining through the soil is known as throughflow. Throughflow from an upslope can fill up the soil pores at the bottom of the slope, and so the soil becomes saturated (Fig. 1). Any extra water is then forced out onto the surface to become OLF. This water, known as 'return flow,' is a component of saturation-excess OLF. Therefore saturation-excess OLF is more likely to occur at the bottom of a hillslope or on shallow soils where there is restricted pore space for water storage.

The area of a catchment or hillslope that produces saturation-excess OLF varies through time. During wet seasons, for example, more of a catchment or hillslope will be saturated and therefore can generate saturation-excess OLF than during dry seasons. During a rainfall, if the catchment starts off relatively dry, then not much of the area will generate saturation-excess OLF; but as rainfall continues, then, more of the catchment becomes saturated, especially in the valley bottoms, and therefore a larger area of the catchment will produce saturation-excess OLF. The fact that the area of a catchment in which saturation-excess OLF occurs tends to vary is known as the 'variable source area concept' (4). The variable source

area model has become the dominant concept in catchment hydrology (2).

Saturation-excess OLF is a much more important process than infiltration-excess OLF across temperate zones. The main differences between the two OLF types are related to the water flow paths. For infiltration-excess OLF, all of the flow is fresh rainwater that has not been able to infiltrate into the soil. However, saturation-excess OLF is often a mixture of water that has been inside the soil (return flow) and fresh rainwater that reaches the hillslope surface.

THROUGHFLOW

If water infiltrates into soil, several things can happen; it can be taken up by plants and transpired (or be lost from the soil by evaporation); it can continue to percolate down into the bedrock; or it can travel laterally through the soil or rock—this is called throughflow.

Most water reaches rivers worldwide either by throughflow, through the soil layers or through bedrock. Throughflow can maintain low flows (baseflow) in rivers by slow subsurface drainage of water and can also contribute to peak flows (storm flow) by generating saturation-excess OLF and as an important process in its own right (5). There are different ways water can move through soil as throughflow, and this impacts the timing of water delivery to the river channel. Soils are not uniform deposits; they have cracks and fissures within them. Water can move through the very fine pores of soil as matrix flow, or it can move through larger pores called macropores (macropore flow), or even larger cavities called soil pipes (pipe flow). Water that moves through the soil matrix occurs in a laminar fashion whereas flow within macropores and pipes is turbulent.

Matrix Throughflow

It is possible to measure throughflow and overland flow on hillslopes by digging a trench across the slope and intercepting and collecting the flow from different soil layers. However, this is not always feasible, and disturbance of hillslope soils is not desirable. Flow through the matrix of a porous substance such as soil that occurs in a laminar fashion should behave according to Darcy's Law. This allows us to calculate the likely rate of water movement through a porous medium when it is saturated (saturated hydraulic conductivity). Thus, it is possible to estimate the amount of flow taking place as matrix throughflow. Often, the saturated hydraulic conductivity will vary with depth and soil type. Sandy soils typically have high hydraulic conductivity compared to clay soils that have low hydraulic conductivity. Therefore, water will drain through sandy soils more quickly than through clay soils. Lateral throughflow through the soil matrix will occur in any soil in which the hydraulic conductivity declines with depth. If both soil and bedrock remain permeable at depth, however, then percolation remains vertical and little lateral flow can occur; infiltrating water will only recharge groundwater storage (5).

Water also moves through soils on hillslopes that are unsaturated. Even after a long drought, most soils contain

some water. This suggests that gravitational drainage and evapotranspiration are not the only forces at work in moving water within soils and that other forces involved must be very strong (2). It is possible to calculate how the interacting forces of gravity, matric potential (soil water tension), positive pore water pressure, and other forces that operate balance out. Once this soil water energy balance has been determined (by measuring pore water pressure and relative altitude, for example), it is possible to map spatially the directions of subsurface water movement. Water in different parts of the soil can move in different directions due to capillary, gravitational, and other forces. Often, it is possible to show that throughflow water preferentially flows into hillslope hollows, and thus these areas become more important contributors to subsurface flow and saturation-excess OLF.

Macropore and Pipe Throughflow

Macropores are pores larger than 0.1 mm in diameter, and soil pipes are those larger than 1 mm in diameter that transport water through the soil. Macropores and pipes can promote rapid, preferential transport of water and chemicals through the soil due to their size and also because they are connected and continuous across sufficient distances to bypass agriculturally and environmentally important soil layers (6). Water may drain more quickly through macropore and soil pipe networks than through the soil matrix. Darcy's Law does not provide a good estimate of subsurface drainage rates if a soil contains many hydrologically active soil pipes or macropores because it is valid only for nonturbulent flows. Macropores may not take up much space in the soil, and if they are open at the soil surface, they will often take up only a tiny proportion of the soil surface. Despite their small spatial role, macropores can still have a high impact on runoff and play a large role in throughflow drainage as water can preferentially flow through them. A study in Niger on a crusted sandy soil showed that 50% of infiltrated water moved through macropores (7). Some studies in upland peat catchments have indicated that 30% of throughflow moves through macropores (8) and 10% through soil pipes (9).

Groundwater Flow

Groundwater is water held below the water table in soil and rock. Therefore, groundwater flow has, to some extent, already been discussed before. However, further treatment of groundwater flow as a separate component is required because of its worldwide importance. In many catchments, water is supplied to the stream from groundwater in the bedrock. This is water that has percolated down through the overlying soil and entered the bedrock. Rock has small pores, fractures, and fissures. Therefore, it is possible to use Darcy's Law to investigate flow rates through bedrock where fissures are at a minimum. Where there are large fractures such as in cavernous limestone areas, then it may not be so useful. Groundwater may be a large store of water, but for it to be available to supply river flow, the holding material (rock or soil) needs to be not just porous but permeable. That is to say that a rock (or soil)

may be porous but relatively impermeable either because the pores are not connected or because they are so small that water can only be forced through them with difficulty. Conversely a rock that has no voids except one or two large cracks will have low porosity and therefore a poor store of water, but because water can pass easily through the cracks, the permeability will be high (5).

RUNOFF PROCESS CONTROLS OF RIVER FLOW

Surface and subsurface flow processes are important controls on river flow and the response of a catchment to precipitation. The dominance of different types of runoff processes is controlled by local climate and catchment features such as geology, topography, soils, and vegetation. Land management can influence runoff processes by changing the infiltration capacity of the soil surface, altering the internal structure of a soil, or by changing the catchment water balance (e.g., through deforestation). Depending on the nature of the aquifer, baseflow discharge may be uniform throughout the year, or peak discharge may lag significantly behind precipitation inputs (2). Some catchments lack any significant groundwater storage and therefore have limited baseflow.

The occurrence of hillslope flow processes in the catchment and their relative dominance affect the speed at which water is delivered to a stream. Overland flow is typically much faster than subsurface drainage. This is why urbanization can lead to increased flood risk downstream. Surfaces created by urbanization reduce infiltration capacity and promote the formation of infiltration-excess OLF. Matrix flow is slower than macropore flow which, in turn, is a form of drainage slower than pipe flow. Many soil pipe flow waters can have velocities equal to that of overland flow.

Where infiltration-excess overland flow dominates the hillslope runoff response, then a river hydrograph is likely to have a short lag time and high peak flow. If throughflow in the small soil pores (matrix flow) dominates the runoff response on the hillslopes, then the hydrograph may have a much lower peak and longer lag times. However, throughflow contributes to saturation-excess OLF so throughflow can still lead to rapid and large flood peaks. In some soils, only a small amount of infiltration may be needed to cause the water table to rise to the surface. There may even be two river discharge peaks caused by one rainfall, where the first peak is saturation-excess overland flow and some precipitation directly in the channel and the second peak is much longer and larger caused by subsurface throughflow accumulating at the bottom of hillslopes and valley bottoms before entering the stream channel. Throughflow may also contribute directly to storm hydrographs by a mechanism called piston or displacement flow. This is where soil water at the bottom of a slope (old water) is rapidly pushed out of the soil by new fresh infiltrating water entering at the top of a slope. Determination of whether the water is old water being pushed out of a slope or new water can be done hydrochemically (by comparing the water to precipitation chemistry) or by using dyes. Where surface saturation occurs to any great extent such as on wide valley bottoms,

saturation-excess OLF will dominate the flow response with higher peak discharges and lower lag times than are characteristic of throughflow storm contributions. Where soils are fairly impermeable through much of their profile, surface saturation may be extensive, and much more rainfall is translated into runoff. Where permeable soils overlie impermeable bedrock, however, throughflow can sometimes account for most storm discharge (5).

Runoff processes are by no means independent of one another, and water travelling across the surface at one point may later take the form of subsurface flow through the matrix and then flow through macropores before being returned to the soil matrix, for example. Hillslope runoff is dynamic, and the dominance of runoff production processes varies across both time and space.

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TRACE ELEMENTS IN WATER, SEDIMENT, AND AQUATIC BIOTA—EFFECTS OF BIOLOGY AND HUMAN ACTIVITY

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INTRODUCTION

Trace elements are defined here as those elements that generally occur in water at concentrations of less than 1 mg/L.

In recent years, the role of trace element studies in environmental chemistry is gaining attention. The

deficiencies or excess distribution of these in water and foodstuffs may pose serious regional health hazards and cause diseases in human beings and animals. Proper evaluation of the environmental impact of trace elements on humans, animals, plants, and aquatic biota requires a multidisciplinary approach involving physicians and geoenvironmental scientists.

TRACE ELEMENTS IN WATER

Trace elements enter the aquatic environment in small soil particles that are redistributed during runoff as suspended particles in streams, lakes, and coastal marine locations. Trace elements in water are important; they are toxic to humans and aquatic life in excess, and they accumulate in food chains. Trace element activity in aquatic systems depends on the element species formed by natural hydrodynamic, chemical, and biological forces.

The concentrations of trace elements in water are often much lower than would be expected from either equilibrium solubility calculations or supply to water from various sources. The most common reason for the low concentrations is adsorption of elements onto a solid phase.

To understand the behavior of any trace elements in waters, it is essential to know the chemical forms of the element in the water. Anomalously high concentrations are often related to the presence of stable complexes in solution (1).

Adsorption of manganese and iron oxides, in particular, is probably the most important process in maintaining the concentrations of trace elements at levels far below those predicted by equilibrium solubility calculations. In general, elements in solution as cationic species are more likely to be adsorbed than anionic species. An important exception is the strong adsorption of phosphate anions by ferric oxyhydroxides (1).

TRACE ELEMENTS IN SEDIMENTS

The significance of studies of sediments in the context of water pollution was realized in the 1960s. Publications show that trace element studies in sediments began in the United States in 1963, the Netherlands in 1966, the Soviet Union in 1967, the Federal Republic of Germany and Sweden in 1968, Poland in 1969, Hungary in 1970, Australia in 1972, Israel and Japan in 1973, Switzerland in 1974, the United Kingdom in 1975, Yugoslavia in 1976, and so on.

The study of sediments is all the more important because, on one hand sediments are potential carriers of elements from a watershed to the receiving body of water, and, on the other, they play a significant role in the attenuation of toxic trace elements in polluted aquatic environments.

Sediments commonly form major sinks for trace element contaminants released during both past and present geological, industrial, and other human activities. Commonly, trace elements released from industrial activity are adsorbed onto other mineral surfaces (2).

Most trace elements in sediments (mercury, lead, cadmium, zinc, copper, nickel, cobalt, manganese, chromium, and silver) are chemically bound and require heat and low pH to convert them to soluble form. Various physical, chemical, and biological factors (such as the scouring action of water currents, dredging, changes in chemical parameters and in microbial activity) can bring about the redistribution and partial solution of trace elements from sediments.

Clay minerals that comprise a significant part of sediments together with freshly precipitated iron hydroxides, amorphous silicic acids, and organic substances can sorb cations from solution and release equivalent amounts of other cations into solution. However, the behavior of different clay minerals toward the various trace elements is not yet fully understood. It is reported that the exchange capacity increases in the order:

$$\text{Kalonite} < \text{chlorite} < \text{illite} < \text{montmorillonite}$$

The affinity of trace elements toward clay minerals has been established as follows (3):

$$\text{Pb} > \text{Ni} > \text{Cu} > \text{Zn}$$

Further, it has been established that Cr and Cu favor bonding to illite whereas V, Co, and Ni favor bonding to montmorillonite.

A general increase of trace element concentration from coarse to fine-grained fractions has generally been noted. Trace element content in particulate matter is 1000 to 100,000 times higher than the corresponding trace element content in the associated aquatic phase. The presence of a large proportion of heavy minerals results in characteristic enrichment of trace elements, particularly in fine sand-silt fractions. On the other hand, the high proportions of light minerals such as quartz and feldspar tend to have the opposite effect on the sedimentary trace element budget (4).

Dahlberg Model

The variations in the trace element content of stream sediments can be expressed as a function of potential controlling factors by the following model of Dahlberg (5):

$$T = f(LHGCVM_e)$$

where T = resulting trace element concentration, f = function of potential controlling factors, L = influence of lithologic units, H = hydraulic effects, G = geologic features, C = cultural (man-made) influences, V = type of vegetative cover, M = the effect of mineralized zones, and e = error plus effects of additional factors not explicitly defined in the model.

In this model, factors other than C create natural pollution that can be recorded as the background value. However, it is not always possible to distinguish between contamination of industrial or domestic origin and natural pollution.

The trace elements immobilized in bottom sediments do not necessarily remain in that condition but may be

released as a result of changes in the aquatic milieu. These are the main causes of remobilization:

1. changes in redox conditions
2. elevated salt concentrations
3. lowering of pH
4. increased use of natural and synthetic complexing agents that can form soluble metal complexes
5. microbial activity

TRACE ELEMENTS IN AQUATIC BIOTA

The role of trace elements in aquatic biota has been treated in earlier literature reviews with emphasis on the toxicity of the individual elements. Only since mercury and cadmium poisoning in Japan has the accent been shifted toward investigations dealing with the influence of trace elements on the metabolism of aquatic biota and the ability of the latter to accumulate both essential and nonessential elements. Essential trace elements are used by aquatic biota in only trace amounts, and a slight increase in the concentration in water could lead to a significant element increase in aquatic biota. Experiments on the impact of trace elements on aquatic biota have been carried out for many years (6–9).

Phytoplankton

The relative short life of phytoplankton (a few days to a few months) makes it very susceptible to changes in abiotic parameters. The element concentrations, therefore, can vary considerably within a short period of time. Knauer and Martin (10) established that the respective element concentrations for cadmium, copper, manganese, zinc, and lead underwent comparable changes in phytoplankton in Monterey Bay, California. The cadmium concentrations in the water distinctly decreased whereas at the same time, the cadmium content in phytoplankton increased. A similar observation was made for manganese.

Macroalgae

Macroalgae can also be used as indicators of elemental pollution. Their elemental content is directly related to the elemental concentrations in water. A detailed analysis of bladder wrack (*Fucus vesiculosus*) populations in coastal waters around Great Britain revealed significantly higher contamination in the eastern Irish Sea and North Sea by zinc, iron, manganese, copper, nickel, lead, silver, and cadmium than in other coastal waters surrounding Great Britain. The maximum values in the entire investigation area were 962 ppm zinc, 1517 ppm iron, 190 ppm manganese, 28.4 ppm copper, 18.0 ppm nickel, 9.0 ppm lead, 0.79 ppm silver, and 20.8 ppm cadmium (dry weight).

In the compiled data of macroalgae investigations, cadmium values are generally between 1 and 2 ppm. Copper concentrations vary widely. Minimum content is near 6 ppm. The lower concentrations of lead are between 2 and 3 ppm. Chromium appears in an average concentration of 2–3 ppm. The minimum concentrations

of manganese lie between 6 and 10 ppm. Nickel has a concentration range similar to that of lead. A maximum of 72 ppb has been reported.

Mosses

It has generally been observed that mosses have a particular storage capacity for lead. Lead content of 277.4 ppm was found in *Fontinalis antipyretica* in the Elsenz River (Federal Republic of Germany). Samples taken from the heavily polluted Ruhr River had cadmium concentrations between 0.17 and 13.2 ppm, and the record amount of 13.2 ppm occurred in *Lagarosiphon major* from Lago Maggiore in Italy.

Higher plants

The roots of higher aquatic plants enable them to incorporate additional elements from sediment (interstitial water). Trace element enrichment differs from species to species. High arsenic enrichment was identified in different families of higher aquatic plants from the Waikato River, New Zealand. Maximum amounts up to 0.5 g arsenic per kg of dry mass were recorded. Different amounts of Cd, Zn, Pb, and Cu in the roots, stems, and leaves of different species were also identified. The highest concentrations were found in the leaves.

Zooplankton

High mercury concentration has been reported in zooplankton from different parts of the world. The maximum concentration recorded was 25.21 ppm Hg. The mercury concentrations in plankton samples gradually diminished with distance from the source. Zooplankton samples taken off the eastern coast of United States between Cape Hatteras and Cap Cod had an average mercury content of 0.5 ppm. From South African coastal waters, a maximum of 132 ppm of zinc was recorded. Cobalt varied from 0.004–0.26 ppm.

Trace Elements in Fish

The study of fish muscle tissue is one means for investigating the amount of elements entering humans by food chain enrichment and has therefore been investigated more than other organs. The absolute increase in trace elements in muscle tissue of contaminated fish is often much lower than that in other organs (11). A generally higher species-specific variation in element content cannot, therefore, always be determined by analyzing muscle tissue. Furthermore, it appears that the musculature becomes enriched by trace elements only when contamination is extremely high. In the musculature of the roach (*Rutilus rutilus*) from a section of the Neckar River, a 25-fold relative increase between minimum and maximum cadmium contamination was registered. In the liver, there was a 156-fold increase and in the kidney, a 196-fold increase (12). When contamination is severe, primarily the gills show elevated cadmium concentrations. The scales of Atlantic salmon (*Salmon salar*) and brown trout (*Salmon trutta*) were analyzed by Abdullah et al. (13) to determine element contamination. In part of the

sample, a distinct correlation was established between concentrations in the scales and in the environment.

Of all the elements, mercury is known to be the most toxic. Several cases have been reported of mercury contamination in fish and shellfish. One of the most striking investigations is that of Miller et al. (14), who detected similar mercury concentrations in museum specimens of tuna and swordfish species and freshly caught fish. In the field study, it was found that, on average, 88.9% of the total mercury in fish musculature was methylmercury, irrespective of the weight, length, and species of the fish. The assumption can be made that most of the mercury in the ecosystem is available to fish as organic mercury compounds.

Trace Elements in Aquatic Biota in Sediments

The distribution of elements in the aquatic milieu shows that the highest element concentrations are generally found in bottom sediment. The amount of elements concentrated there present, however, a particular danger to organisms, especially to those that live in the sediment and enter the food chain. Kushner (15) found bacteria in the Ottawa River of which 50% and more were resistant to 1 ppm Hg^{2+} and some were even resistant to 10 ppm Hg^{2+} . These bacteria could transform Hg^{2+} into a volatile mercury compound. The accumulation of radioactive cadmium from sediments and sea water by polychaetes was observed by Ueda et al. (16). Worms in contact with sediments accumulated six times more cadmium than worms not in contact with sediments.

Zn, Cu, Pb, and Cd elemental values in goldfish were much higher than those in catfish. This discrepancy can probably be explained by the different feeding habits of the species; the goldfish burrow when searching for food, whereas the catfish is a predatory fish.

EFFECTS OF GEOLOGY

This is the source of baseline or background levels. It is to be expected that in areas characterized by element-bearing formations, these elements will also occur at elevated levels in the water and bottom sediments of the particular area. Trace elements may be derived from the weathering of rocks. Average concentrations of some trace elements in rocks are shown in Table 1.

Whether a particular trace element goes into solution during weathering depends on the mineral in which the element occurs and on the intensity of chemical weathering. Many trace elements do not substitute readily in feldspars or common ferromagnesian minerals. They may be present in chemically resistant accessory minerals such as zircon, apatite, or monazite, or as sulfides, which weather rapidly in oxygenated water. The resistant minerals generally remain unaltered unless weathering is very intense (gibbsite formation), and so alteration of them never causes high trace element concentrations in water. Where many elements occur in high concentrations and hence in many economic deposits, the elements occur as sulfides (e.g., copper, zinc, molybdenum, silver, mercury, lead), and these sulfides often contain selenium, arsenic, and cadmium. Sulfides weather rapidly, so such ore deposits can give rise to local high concentrations of dissolved trace elements.

EFFECTS OF HUMAN ACTIVITY

Human activities introduce trace elements into the hydrosphere in many ways. Burning of fossil fuels and smelting of ores put elements into the hydrosphere, where they are washed out by rain into surface water.

Table 1. Concentration of Selected Elements in Rocks (ppm)^a

	Granite	Basalt	Shale	Sandstone	Limestone
Aluminum	Major	Major	Major	Major	4200
Arsenic	2	2	13	1	1
Beryllium	3	1	3	—	—
Boron	10	5	100	35	20
Cadmium	0.13	0.2	0.3	—	0.03
Chromium	10	170	90	35	11
Cobalt	4	48	19	0.3	0.1
Fluorine	800	400	740	270	330
Iodine	0.5	0.5	2	1	1
Lead	17	6	20	7	9
Lithium	30	17	66	15	5
Manganese	450	1500	850	50	1100
Mercury	0.03	0.001	0.4	0.03	0.04
Molybdenum	1	1.5	2.6	0.2	0.4
Nickel	10	130	68	2	20
Rare earths	0.5–70	1–80	1–80	0.05–15	0.05–8
Scandium	10	30	13	1	1
Selenium	0.05	0.05	0.6	0.05	0.9
Thallium	1.5	0.2	1.4	0.8	—
Thorium	14	2.7	12	5.5	2
Titanium	Major	Major	Major	Major	400
Uranium	3	1	4	2	2
Vanadium	50	250	130	20	20

^aReference 17.

A wide range of trace elements in fossil fuels are either emitted into the environment as particles during combustion or accumulate in ash which may itself be transported and contaminate water. Some of the elements arising as pollutants from fossil fuel combustion are Pb, Cd, Zn, As, Sb, Se, Ba, Cu, Mn, and V.

Municipal sewage and industrial effluent introduce trace elements directly. Mining activities can result in release of elements because previously impermeable rocks are broken up and exposed to water and because sulfide-containing rocks are exposed to oxygen, resulting in rapid alteration and dissolution. In the mining industry, there are two major contaminant waste streams: (1) elements discharged in solution via mine drainage and (2) particulate grains of ore-forming or related minerals released after ore processing (18). Underground disposal of toxic wastes, including radioactive wastes, has the potential to release a variety of substances to groundwater and hence to surface waters.

Agriculture constitutes one of the very important nonpoint sources of trace elements. The main sources are

1. impurities in fertilizers: Cd, Cr, Mo, Pb, U, V, Zn, (e.g., Cd and U in phosphatic fertilizers)
2. pesticides: Cu, As, Hg, Pb, Mn, Zn
3. wood preservatives: As, Cu
4. wastes from intensive pig and poultry production: Cu, As
5. composts and manures: Cd, Cu, Ni, Pb, Zn, As

The human input of trace elements in many rivers and lakes is many times greater than the natural input.

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INNOVATIVE PENS HATCH THOUSANDS OF TROUT

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For a fellow who doesn't fish, Pittsburgh District's Mike Fowles has sure helped to put a lot of trout into the creels of Pennsylvania anglers.



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A fish and wildlife specialist for the U.S. Army Corps of Engineers for the last 26 years, Fowles has worked out a landmark agreement with a local chapter of Trout Unlimited to establish a cooperative fish hatchery in the tailrace of the district's Youghiogheny Lake. The effort has put an estimated 20,000 trout ranging from 12 to 16 inches into various area waters, and earned Pittsburgh District recognition across the state as a friend of fish.

The Chestnut Ridge Chapter of Trout Unlimited (TU) this spring signed a five-year extension to its memorandum of understanding (MOU) to run the hatchery. Other signers along with the Corps were the Pennsylvania Fish and Boat Commission and D/R Hydro, operators of an electric power generating plant at the dam. The initial three-year MOU was due to expire in June, and has been hailed as a success beyond anyone's expectations.

The key to this success, according to Fowles, is the use of "cage culture" nursery pens actually submerged in the flowing water of the Youghiogheny River, rather than the more common hatchery practice of long concrete tanks with artificial flows and aeration.

The fish arrive in early summer as three-inch fingerlings, and are fed a high-quality protein meal by TU members, who are also responsible for trash pick-up and maintenance of the submerged pens. Corps and D/R Hydro monitor water quality in the tailrace to insure that the young trout have the best possible living conditions. The result a year later is that thousands of the former fingerlings are more than a foot long, and ready to be stocked. "This is a really good growth rate," said Fowles.

Some of the fish go right into the Youghiogheny River (one of Western Pennsylvania's premiere trout streams) while others are trucked to other nearby waters. The in-river hatchery offers much better growth and survival rates for the trout, which include rainbows, brown trout, brook trout, and an exotic strain of rainbows known as "palomino" trout.

Sections of the river downstream from the hatchery are designated "trophy trout" sections by the State Fish and Boat Commission, and draw anglers from throughout the eastern U.S.

The state provides the trout fingerlings each spring. For its part, the hydroelectric company provides free power from its nearby generators to support the hatchery.

"This is truly a win-win situation for all the partners," said Fowles on the day of that the partners extended the MOU. "This is an outstanding example of how Pittsburgh District is developing new partnerships to accomplish our natural resource management and environmental stewardship objectives."

Tom Shetterly, president of the Chestnut Ridge Chapter of TU, said about a half dozen of his group's members rotate the daily feeding duty for the young trout through the year. "When it's time to stock, we get 12 to 15 guys there," he said. The annual feeding bill runs about \$2,000.

He said that the Corps of Engineers "is just a great partner with us; you guys have just bent over backwards to make this program better than anybody could have ever expected." He also praised the efforts of other Pittsburgh District personnel, including Mike Koryak,

district limnologist (one who studies fresh waters), and Pat Docherty, area operations manager.

Working with the Corps and the state, TU has also received permission to place gravel for enhanced natural spawning beds in the river and the stilling basin below the Yough Dam.

"We're seeing some of the first fish we stocked come back to the same area to spawn," Shetterly said. Raising the fish in submerged cages apparently has imprinted them with the characteristics of their immediate surroundings, something that Shetterly says has never been accomplished elsewhere. "This thing is huge," he marveled. "It's going to be world-famous."

Another bonus of the in-river cage culture is the way the trout benefit from natural food, adding up to 12 ounces in weight gain for each pound of artificial fish food added to the cages. Waste from the trout and any food that goes through the bottom of the cages also combine to increase the alkalinity of the water, thus improving the environment, according to Shetterly.

The trout hatchery is one of many programs supporting fishing in Pittsburgh District. For example, working with the Yough Walleye Association, the district is supporting efforts to put crushed limestone into Tub Run, a tributary of Youghiogheny Lake, to neutralize the highly-acidic waters, a legacy of abandoned coal mining.

Youghiogheny Lake is one of 16 multi-purpose flood control reservoirs built and operated by Pittsburgh District in Western Pennsylvania, New York, Ohio, West Virginia, and northern Maryland. Pittsburgh District also works with various state fish and wildlife agencies to conduct annual fish census sampling at its navigation locks on the Ohio River.

All in all, Fowles has worked hard to help improve fish habitat and fishing as a sport, but don't ask him to tell you his own fishing stories. "I don't fish," he said simply. "And I don't hunt either, though of course I recognize both as legitimate outdoor activities."

He is passionate about bird watching, a hobby he and his wife picked up in 1973 while an undergraduate at Ohio University. His logbook shows he has sighted thousands of bird species all over the world.



Thousands of brook, rainbow and brown trout are raised each year in these submerged pens in the tailwaters of the Youghiogheny River. (Photo courtesy of Pittsburgh District)

"My list is rapidly approaching the 5,000 mark, a goal I set for myself when I first started birding." Comparing the two sports, he says of birders, "Oh, we get our prey. We just don't kill 'em."

WATERSHED

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DEFINITION

Everyone lives in a watershed. A watershed is the hydrologic or geographic boundary of a river system. Watersheds can be delineated for any waterbody, whether it is the creek flowing through your neighborhood or the Mississippi River (Fig. 1).

Watersheds are defined by first selecting a point along a stream or river. The stream and its tributaries, if any, are then traced backward from there, and the topography of the terrain is analyzed. The watershed boundary is defined by the high point, or ridge, between streams flowing to separate major waterbodies or "stems." All waters falling within the watershed boundary, in theory, eventually flow to the original point designated to define the watershed.

WATER QUALITY

All factors and activities within a watershed directly or indirectly affect the water quality of river systems. Although local climate, geology, soil type, and vegetation play an important role in determining of the water quality conditions within a watershed, one of the most important factors affecting water quality is land use. Runoff from different land use types, such as agricultural, industrial, urban, or even barren or deforested land may contribute a number of pollutants to a waterbody. Some of these pollutants include excessively applied fertilizers or pesticides, animal waste, toxic chemicals, sewage from broken sewer lines or septic tanks, oil, and sediment.

Best management practices on all types of land use help reduce the amount of pollution entering waterways. Preserving, restoring, or creating wetlands, which "filter out" many pollutants is another way to help ensure clean water and a clean environment within a watershed.

FLOOD POTENTIAL

Flooding is also influenced by factors such as the percentage of impervious cover within the watershed boundary. In addition, concrete drainage control ditches and canals have been built over the years to help control flooding. When a channel is lined with concrete, natural aquatic habitats are disturbed or destroyed, and pollutants

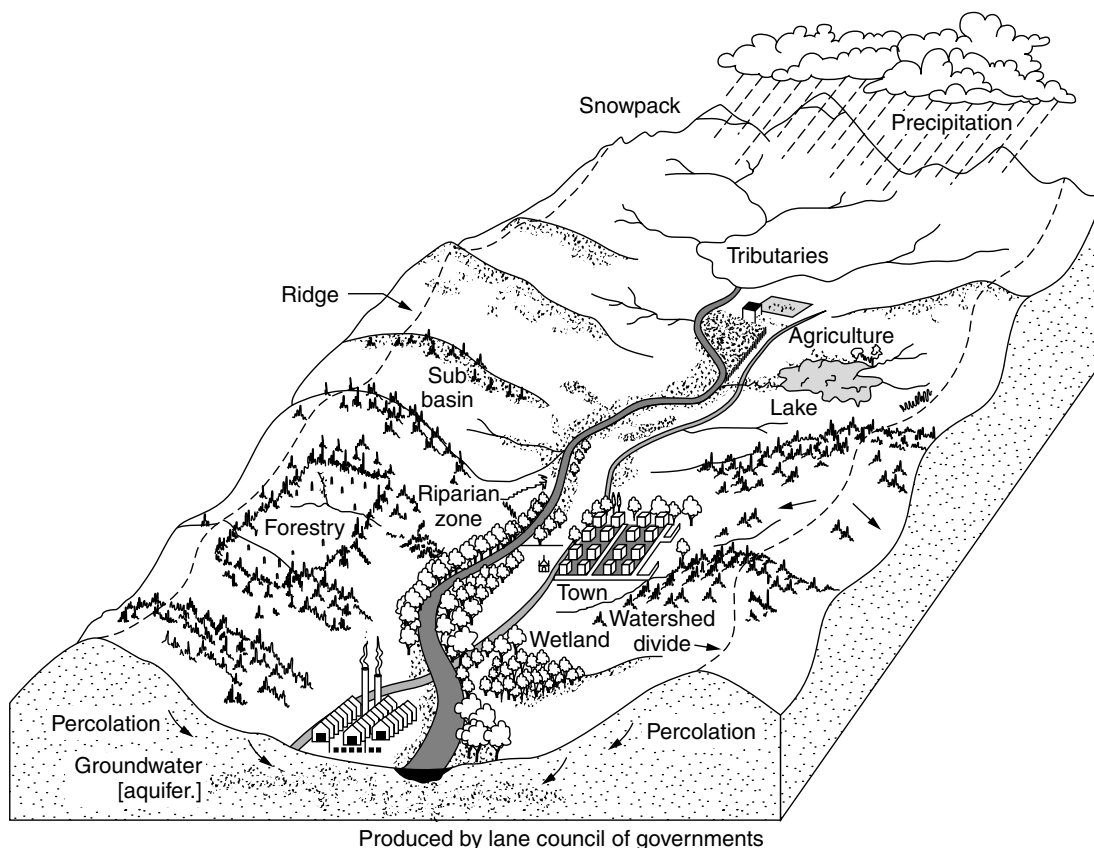


Figure 1. Sample drawing of a watershed (Used with permission of the Lane Council of Governments, Eugene, Oregon).

can more easily enter a waterway due to the lack of vegetation to control and filter runoff.

Recently, engineers have used more "environmentally friendly" types of flood control. Some these include constructed wetlands and detention ponds. In these cases, habitat is not destroyed, but created, and pollutants are kept from entering waterways.

Although a person or entity may not be located near a waterbody, the results of various actions eventually flow downhill or through stormwater infrastructure to creeks and streams.

COMBUSTIBLE WATERSHEDS

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Existing hydrological networks can be used for conflagration prediction. Runoff data allows predicting high fire danger earlier, highlights the most dangerous watersheds, and makes forecasting more foolproof. In some cases, a hydroforecast improves the likelihood of preventing a major disaster. This technique has certain differences from standard meteorological methods: (1) It predicts conflagrations not for an administrative or geographical region, but for a watershed—so, a conflagration can be forecast for the area around water supply reservoirs. (2) It allows forecasting the fire danger earlier and estimates of the amount of time available for active response. (3) It excludes most false alarms. (4) It is more accurate, simply because it uses hydrographic data, which are more

representative than meteorological data. (5) It is more foolproof and less dependent on errors in observations. (6) It depends less on the stochastic nature of summer rains and other meteorological elements. (7) It is simpler than any other method.

THE SIMPLEST TECHNIQUE

1969

Watershed = 671 sq.km, 30 km SSW from Canberra City (capital of Australia), adjacent to the water supply watersheds of Cotter River (mainly eucalyptus forest overlying granitic soils).

1970

45° = baseflow recession during cool periods (semilog scale for cumecs).

1971

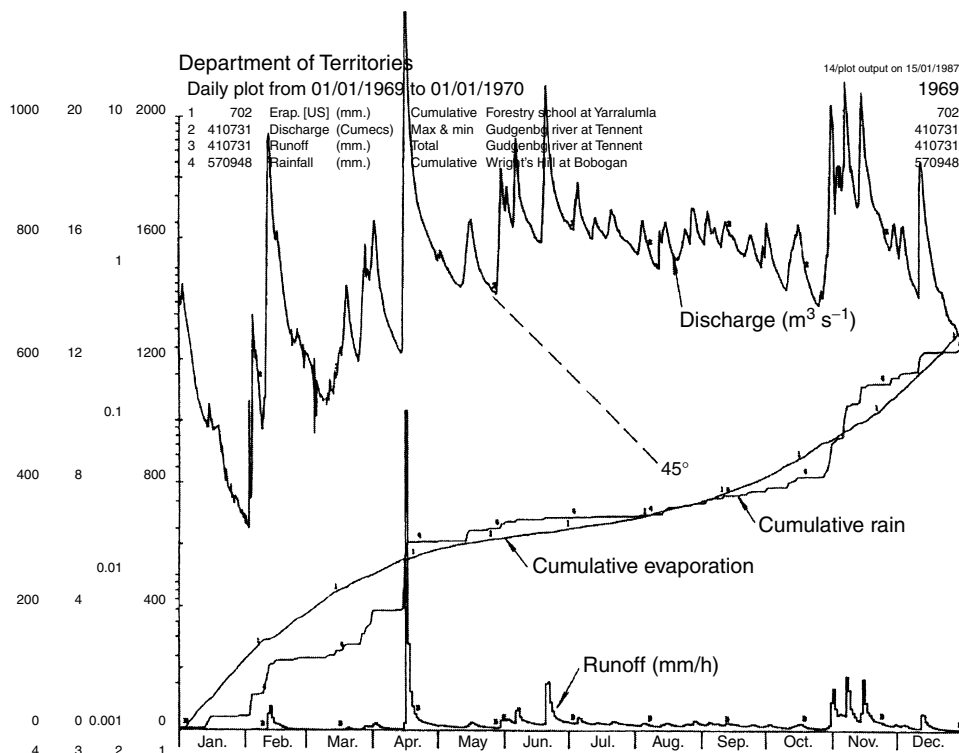
Vertical scales have to be the same for all years.

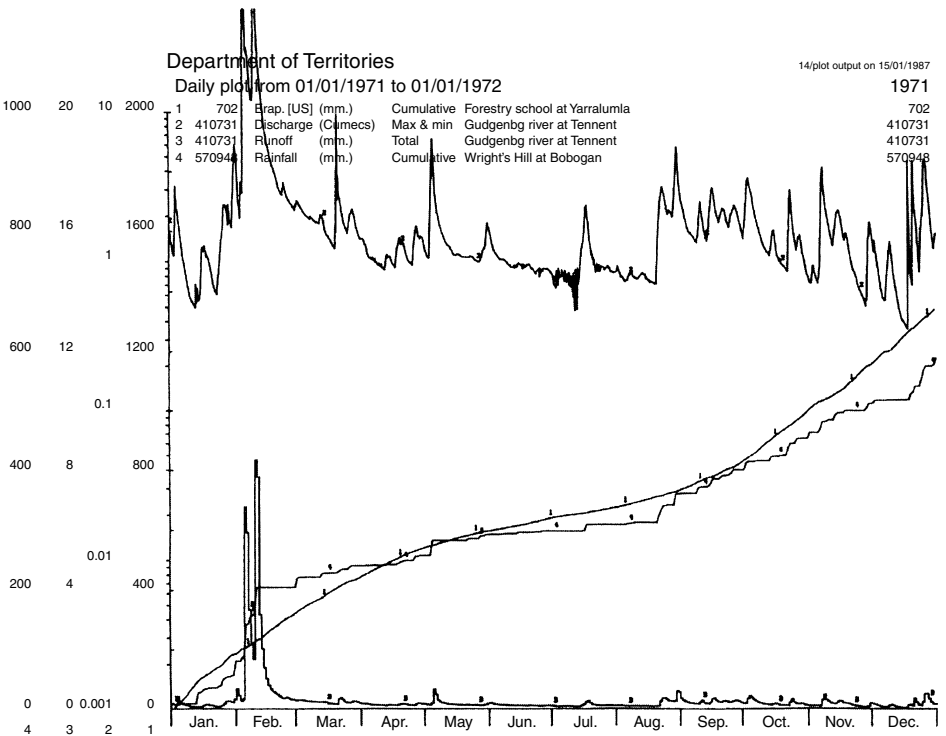
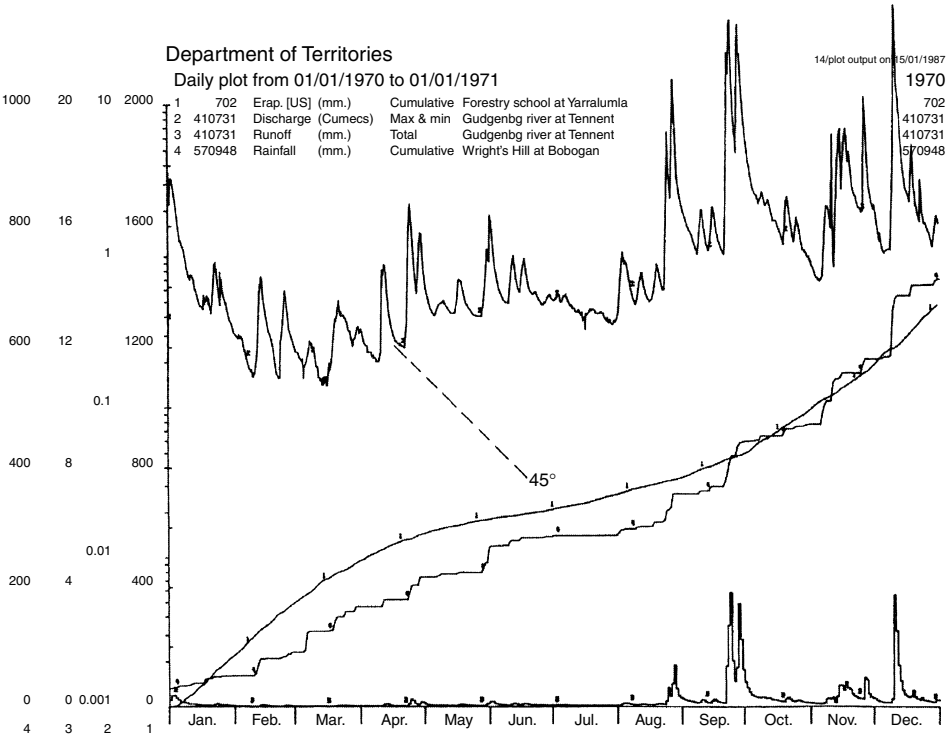
1972

Every summer there are many small fires. A fire > 1 sq. km starts, when the discharge < 0.1 cumecs and the recession > 45° .

1973

For this and 4 previous years: cumulative evaporation = two cumulative rainfalls.



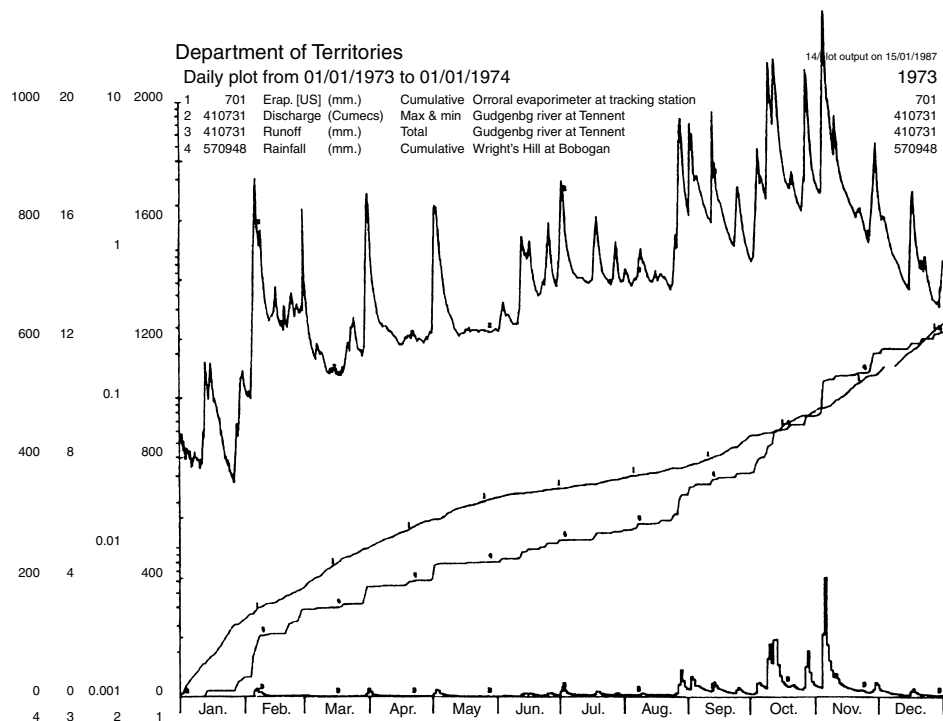
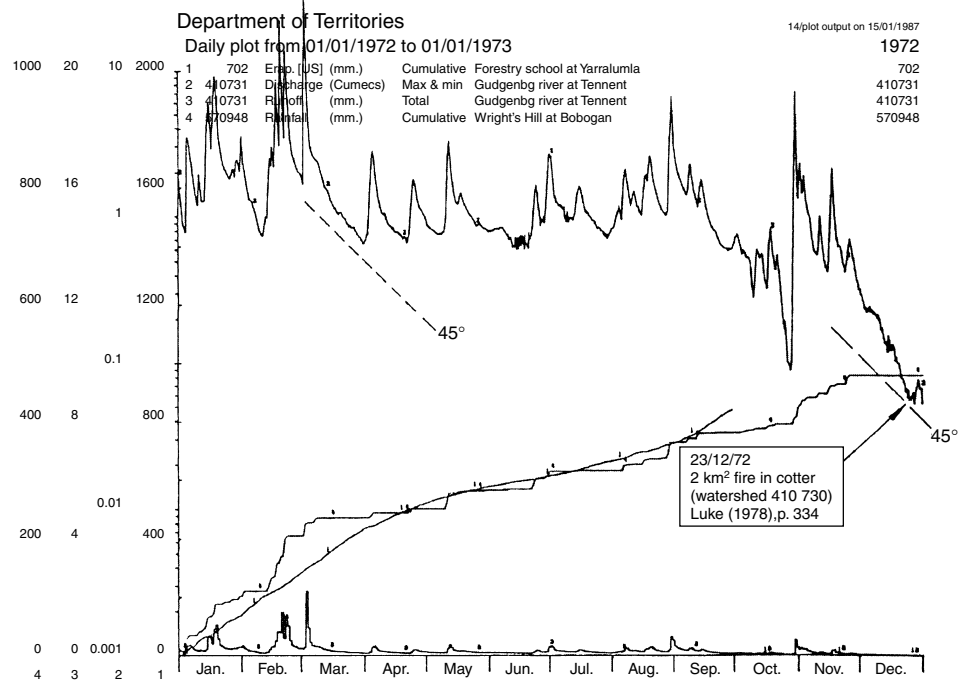


1974

Floods: Cumulative evaporation < two cumulative rainfalls. Meteo database problem: “negative rain” for October 1974. Dryness of fuel affects the intensity of a fire. Several special indexes are used to forecast it. There are no approved international standards for calculating these

indexes, and even in one country, several different drought and fire danger indexes may be used (1–2).

The idea behind any antecedent moisture content index is to “sum” the rainfall and evaporation across a certain time interval. Hence, a mistake in a daily rain record could be fatal, not only for this daily index, but also for subsequent days.



1975

Small fires only. Defect of meteo data: evaporation. Turner (2, p. 12) stated that "a single day of careless weather observations could strongly affect fire weather indices."

1976

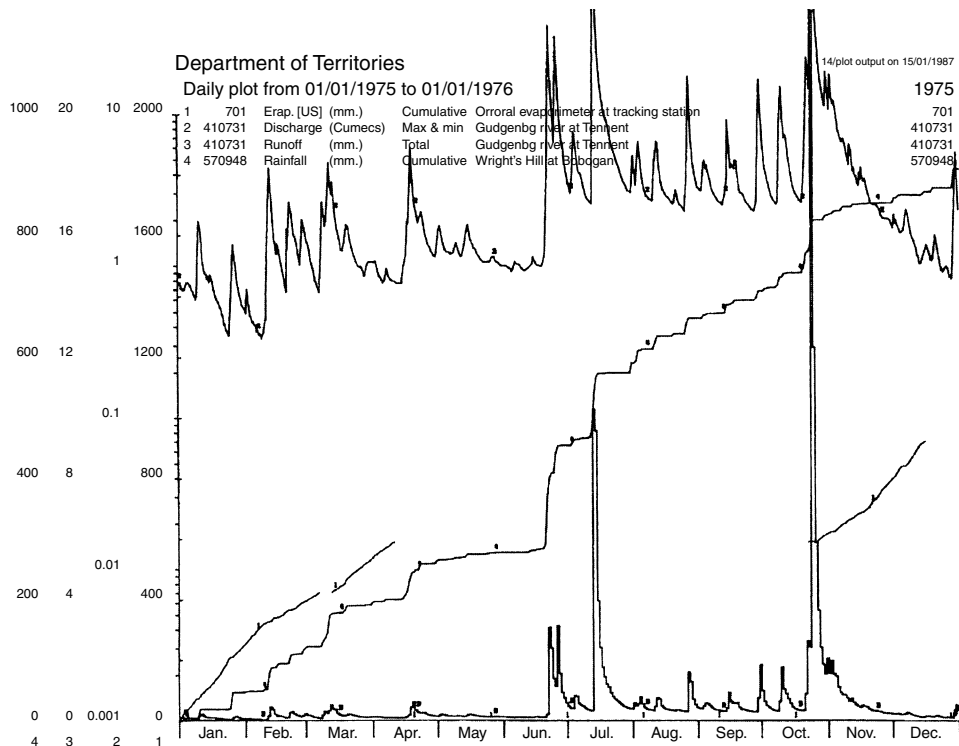
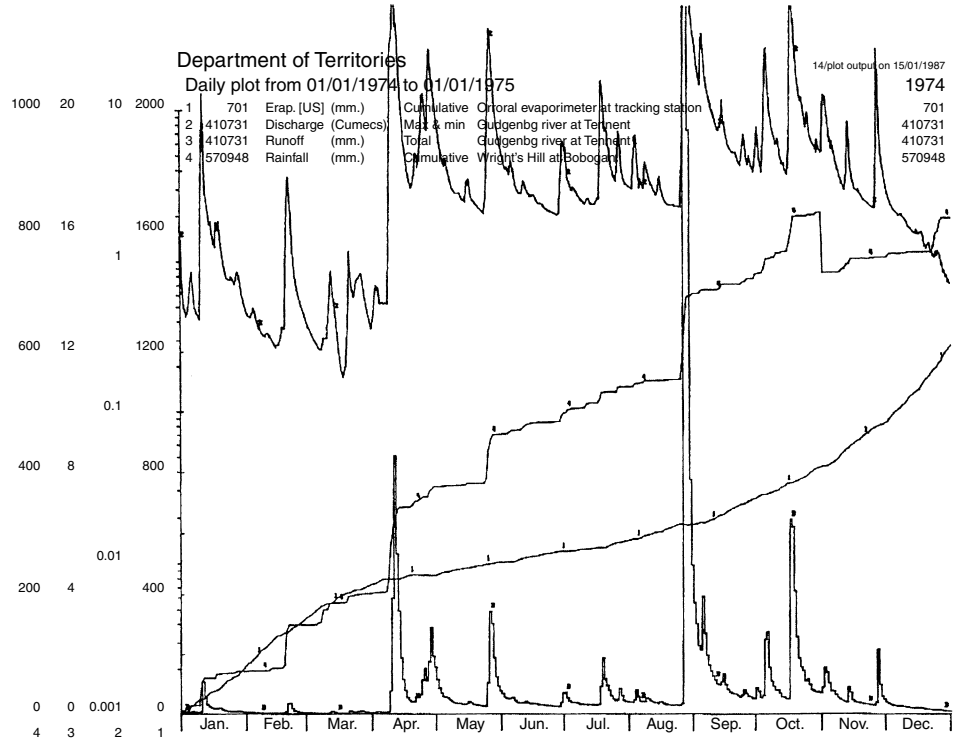
Similar problem with meteo data (as in 1972, December 1973, 1975).

1977

Low flow does not indicate a fire, if the recession = 45° (compare with 1972).

1978

February: discharge = 0.1 cumecs, but the recession is concave. Small fires only.

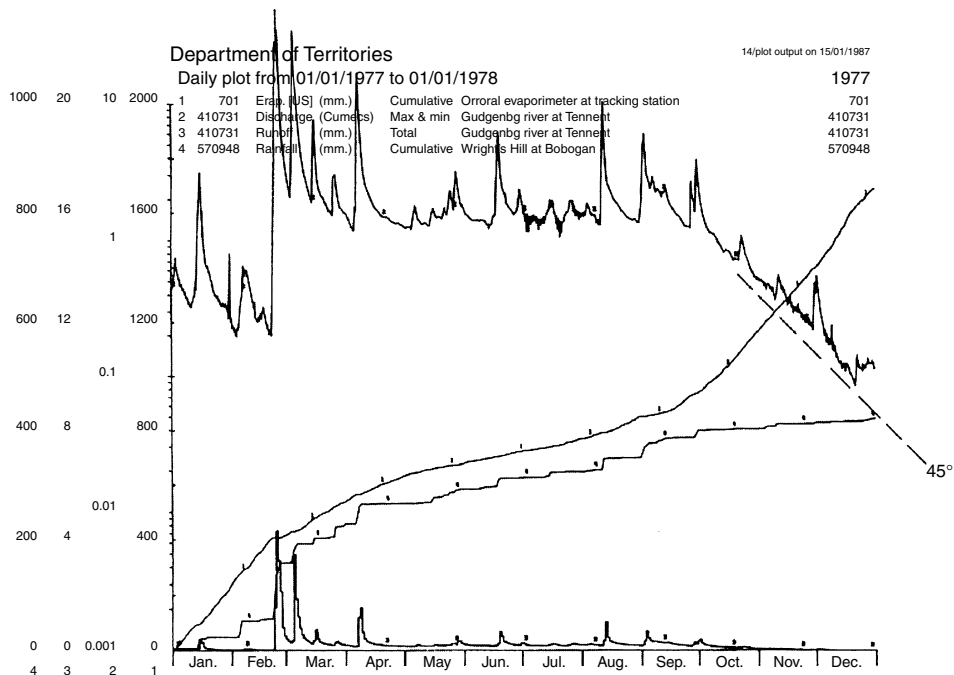
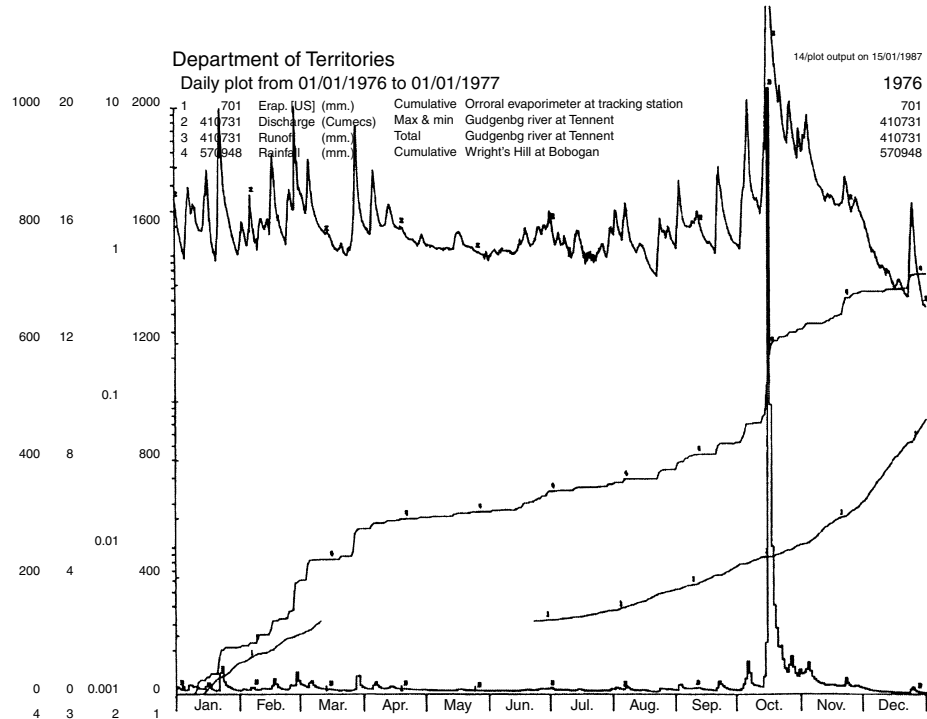


1979

Small fires only. Cumulative evaporation > two cumulative rainfalls, meteo fire danger is "extreme." It does not mean, however, that the meteo indices and other techniques of bushfire forecasting are to be abolished and forgotten. The hydrologic technique only supplements them.

1980

Cumulative evaporation > two cumulative rainfalls, but the recession is concave. Small fires only. Bushfires can start almost any time. Even during autumn rains, lightning could start a bushfire. A wet forest will not burn as well as a dry one, but it may burn. However,



the intensity of such bushfires is usually very low, and it is generally thought that such bushfires are not a major hazard. They decrease the amount of the accumulated fuel and hence to some extent, prevent the more dangerous, intense conflagrations.

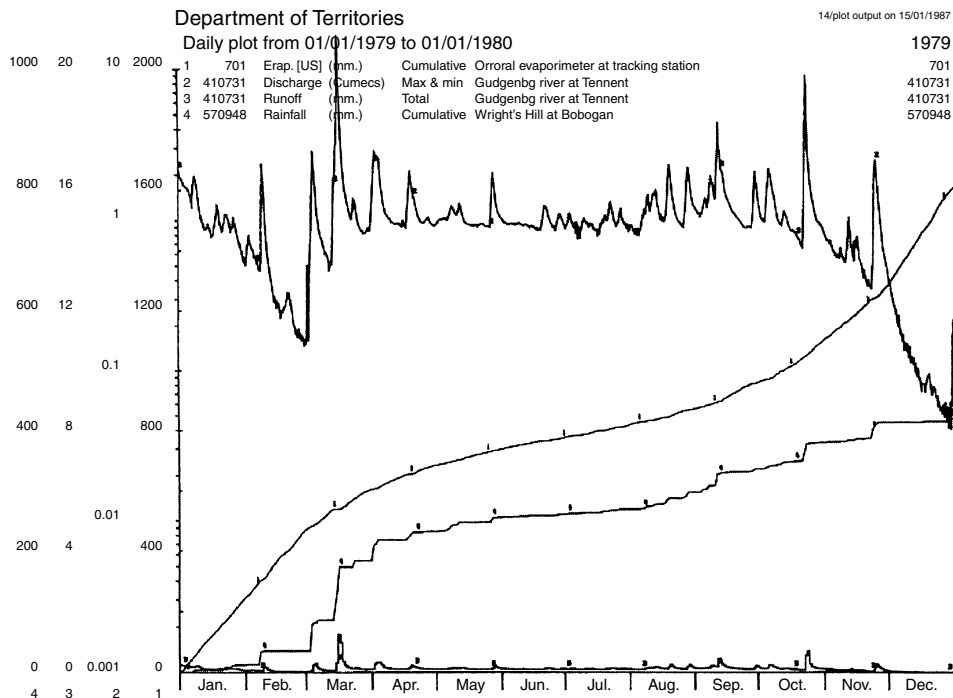
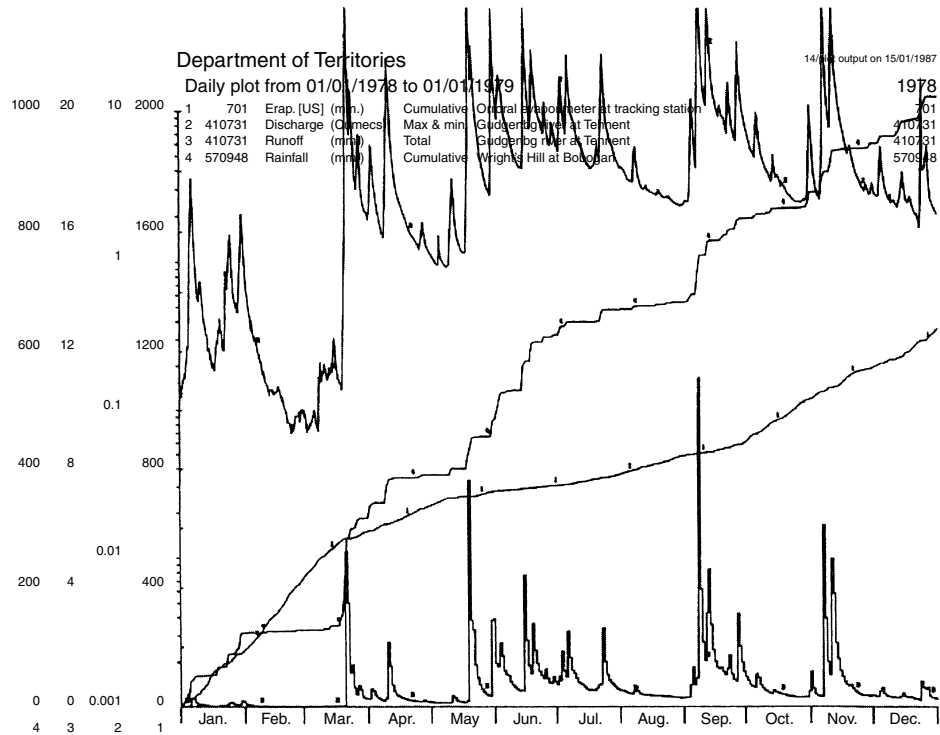
1981

Cumulative evaporation > two cumulative rainfalls, but the recession is concave. Small fires only. Small fires are used for prescribed burnings and are considered generally

a useful tool when controlled effectively and sufficiently, little damage results in the long-term components of the environment. These artificial bushfires are called Fuel Reduction Burnings (3–5). The 1981 hydrograph shows that it was a good year for prescribed burnings.

1982

There are two groups of predictors: (a) Early predictors: (a1) low flow; (a2) change of the curvature of the recession curve from concave to convex (semilog paper); (a3)

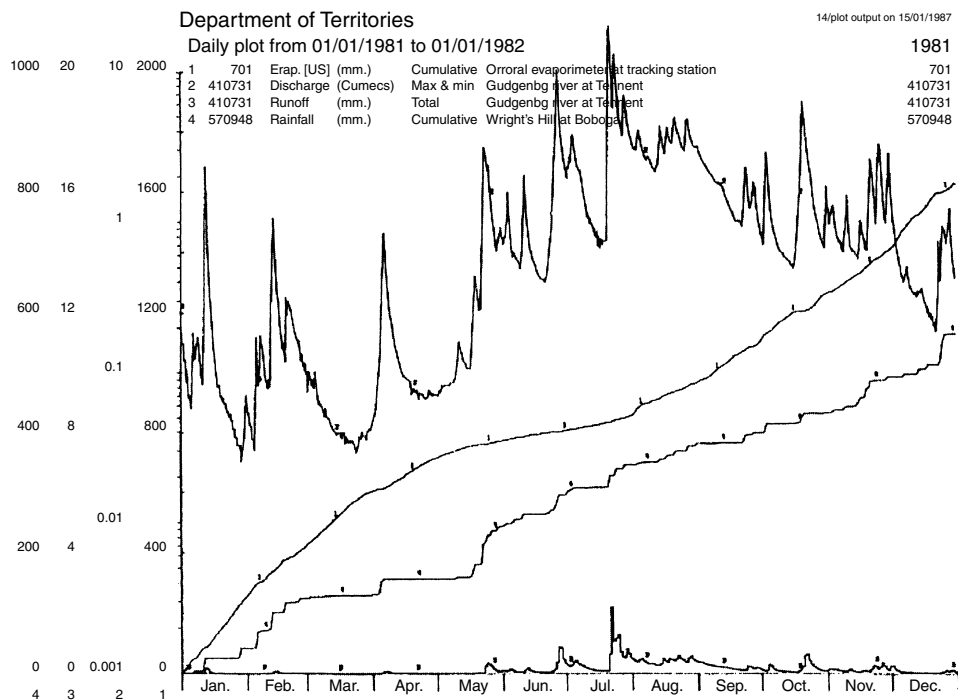
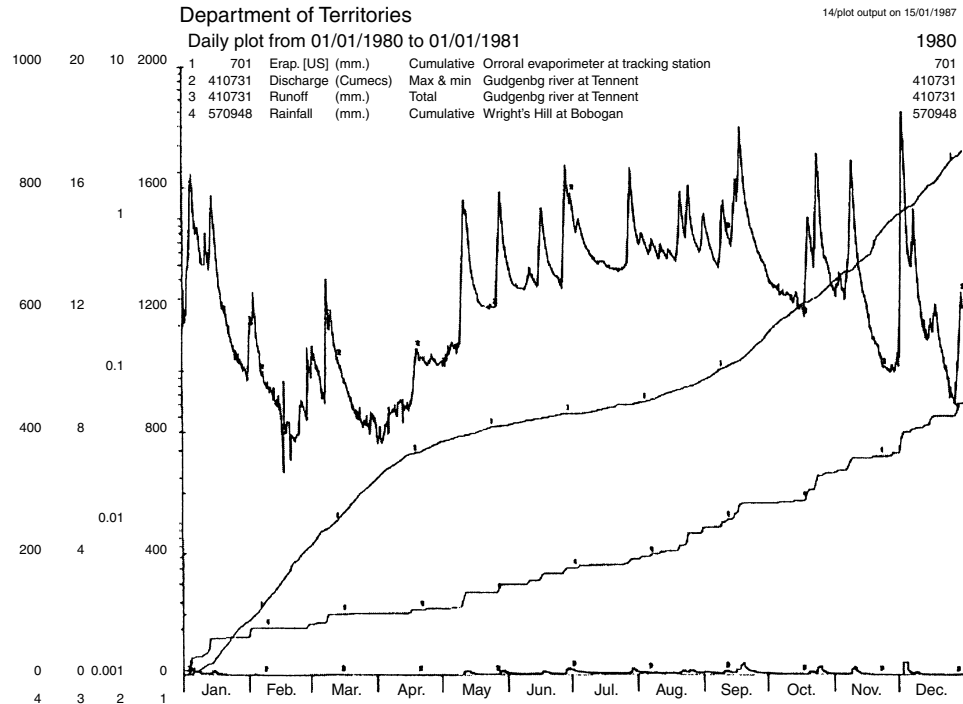


accelerated recession. These early predictors can be used to estimate how many days there are before the critical fire danger starts (by extrapolating the recession curve). (b) Later predictors: (b1) stabilization of the runoff to a "zero flow" value; (b2) Steep rate of recession after intermediate minor rains; (b3) Convex recession after intermediate rains. (b4) Abrupt stabilization of runoff to "zero flow" several days after a minor rain. The manifestation of

these late predictors means that the probability of a conflagration has increased dramatically.

1983. Combustion January 9, 1983

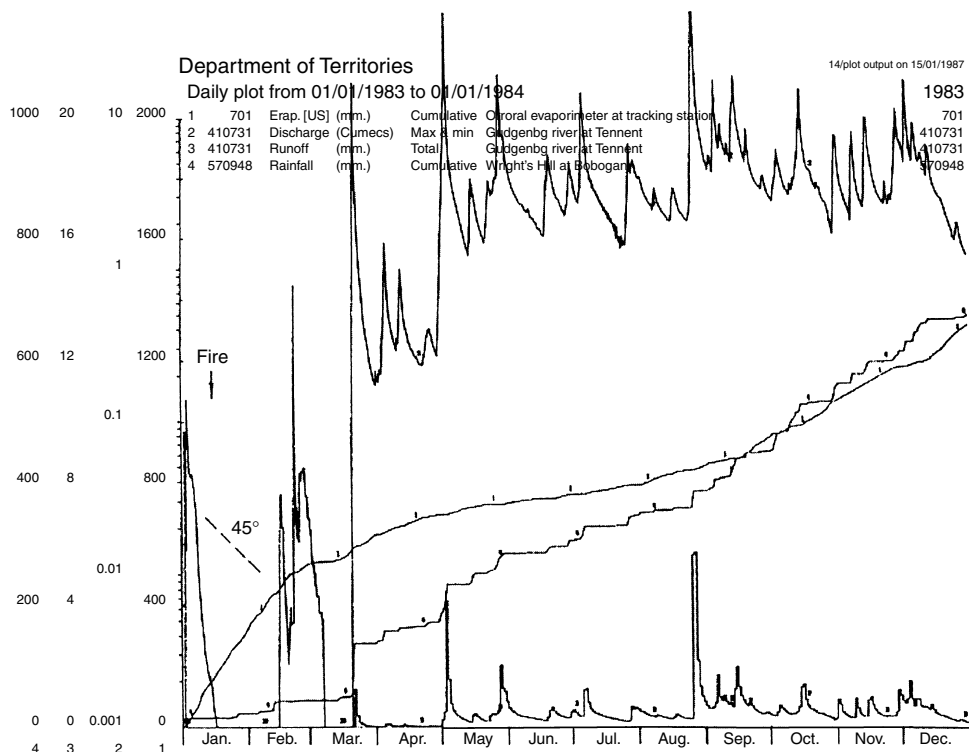
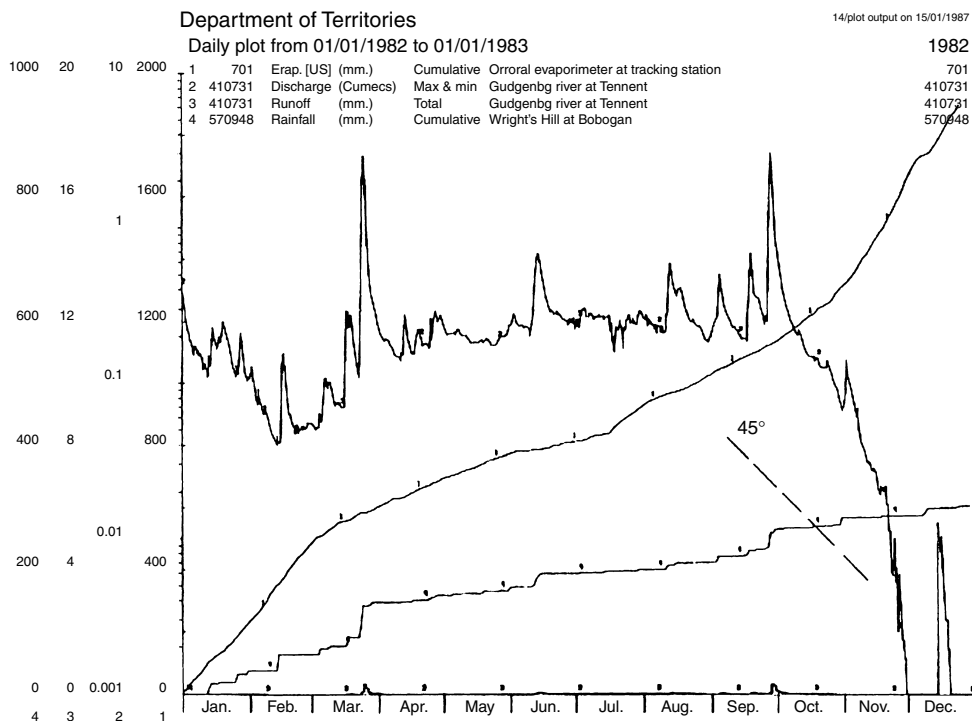
In January 1983, a wild fire burnt out approximately 58 km², representing 30% of the Corin Dam catchment on the Cotter River and 300 km² of the adjacent Gudgenby River catchment. It was a conflagration of



very high intensity. The conflagration completely burned the Boboyan pine tree plantation (300 ha), despite all attempts to save it (by concentrating firefighting resources on protecting the plantation). Later on, attempts to control the conflagration were only partly successful, as the conflagration continued to spread for about 3 weeks. (Note recession after intermediate rains). The approximate cost of conflagration suppression operations was \$477,000 (6).

SUMMARY

A major forest conflagration is a very rare phenomenon; to collect a sufficient amount of information for development of the forecasting technique, it is necessary simply to analyze more cases. We ask everybody who has at hand data on watersheds, affected by a conflagration, to try this simple technique, to develop it, and to communicate their results.



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TIME OF CONCENTRATION AND TRAVEL TIME IN WATERSHEDS

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INTRODUCTION

Time of concentration, denoted by T_c throughout this article, is one of the oldest and most fundamental concepts in hydrologic science and engineering. Time of concentration and travel time (T_t) are two measures of watershed runoff response time. Travel time represents a broader hydrologic term, and time of concentration may be considered a special case of travel time. Several hydrologic analysis and design procedures require direct knowledge of T_c or T_t . In the rational method, T_c must be known to determine the duration of design storm and thus, the flood peak. The Muskingum stream routing method requires wave travel time through stream reaches as a measure of the storage time constant. The application of time–area or Clark rainfall–runoff transformation is optimum when the spatial distribution of travel time is known.

DEFINITIONS

Time of concentration is usually defined as one of the following categories:

1. The time that it takes for water to travel from the most distant part of a watershed to the outlet. Some literature appropriately emphasizes that the distance has a hydraulic base and not necessarily a geometric base.
2. The wave travel time from the most hydraulically remote point to the outlet of a watershed.
3. The time elapsed for which all parts of a watershed contribute to direct runoff at the outlet.
4. In hydrograph analysis, T_c is defined as the time between the end of excess rainfall and the inflection point on the falling limb of the hydrograph. It is assumed that the inflection point indicates the end of direct runoff. The centroid of the rainfall hyetograph is sometimes defined as the beginning of T_c .

Definitions 2, 3, and 4 represent a similar interpretation of watershed time response, but definition 1 is based on water velocity which is different from wave celerity. As we will see in the next section, some methods explicitly rely on definition 1 for estimating T_c . Definitions 2 and 3 are similar but depend on the duration of rainfall excess (T_r). If, in theory, excess rainfall intensity is constant and T_r is sufficiently long or at least equal to the time to equilibrium (T_e) at which the watershed reaches a steady state, then $T_c = T_e$. On the other hand, when $T_r < T_e$, definitions 2 and 3 are still valid, but the time in neither is equal to T_e . The case of $T_r < T_e$ represents a dominant case in the hydrology of watersheds, so time of concentration becomes event-dependent, not just intensity-dependent, and this is not convenient in practice. Even almost all hydraulic T_c equations avoid dealing with the issue of variable rainfall duration.

Travel time follows definition 2 and may be determined from any point in the watershed to another point of interest downstream. Thus, time of concentration is specifically a travel time whose starting and ending points are the hydraulically farthest and the outlet of the watershed, respectively. By restricting travel time to channel systems, Linsley et al. (1) describes T_t as the time between a crest at one station and the crest at a downstream station. Again, some definitions of travel time refer to definition 1, in which the water travel time is considered rather than wave travel time. T_t is also a subject in underground flow, but in this article only surface water is discussed.

THEORETICAL CONSIDERATIONS

Uncertainty in computing T_c and T_t representing watershed time responses is due mainly to simplifications of complex rainfall–runoff processes. Watershed time response depends strongly on watershed static characteristics, such as shape, area, slope, length, roughness (neglecting long-term changes), and drainage pattern. On the other hand, time response is a function of non-stationary factors such as spatial and temporal rainfall variations. Therefore, watershed time response is generally event-dependent and not a stationary character of a watershed.

Following definition 2, we first assume:

1. Excess rainfall intensity remains constant through time.
2. Rainfall duration is sufficiently long to reach a state of equilibrium.

The most comprehensive analytical derivation of travel time in watersheds with spatially variable characteristics is presented by Saghafian and Julien (2). The general relation for the travel time from location X_1 to location X_2 downstream is as follows:

$$T_t = \int_{X_1}^{X_2} \frac{dx}{c} \quad (1)$$

where x is the distance measured along the hydraulically longest flow path from X_1 to X_2 and c is the wave celerity.

Skipping details of substitutions and using the Manning resistance equation, one can finally obtain

$$T_t = \int_{X_1}^{X_2} (1 - \gamma) \left(\frac{a_1}{Q_e} \right)^\gamma \left(\frac{n}{a_2^{2/3} S_0^{1/2}} \right)^{1-\gamma} dx \quad (2)$$

where Q_e is the equilibrium discharge, a_1 , a_2 , and γ are flow cross-sectional parameters, S_0 is the bed slope corresponding to the *kinematic travel time*, and n is the Manning roughness coefficient.

The equilibrium discharge Q_e passing through a given cross section at location x is equal to the excess rainfall rate spatially integrated over the drainage area of that cross section. This is mathematically expressed by

$$Q_e(x) = \int_0^{A(x)} i dA \quad (3)$$

where $A(x)$ is the drainage area or flow accumulation at distance x and i is the excess rainfall intensity at any point draining to x . The substitution of Q_e in Equation 2 requires knowledge of the spatial variability of excess rainfall and of the drainage area along the hydraulically longest flow path.

Total travel time (T_t) can be separated into the travel time for overland flow, T_{to} , and for channel flow, T_{tc} , such that $T_t = T_{to} + T_{tc}$. Travel time may be computed for all locations in the watershed using Eq. 2. Note that Eqs. 2 and 3 hold for spatially variable excess rainfall. For $X_1 = 0$ at the most hydraulically distant point and $X_2 = L$ at the outlet of the watershed of interest, $T_t = T_c$. Note that the hydraulically longest path may be unknown at first, so one must determine T_t by trial and error.

Certain relations must be obtained for flow cross-sectional area $A_x(h, x)$, hydraulic radius $R(h, x)$, bed slope $S_0(x)$, and equilibrium discharge $Q_e(x)$, or alternatively i and $A(x)$, so that the travel time integral for channel flow can be calculated. Both A_x and R may be expressed as functions of flow depth h , either precisely through geometric relationships or approximately through regression curves. In a general form, $A_x = a_1 h^{b_1}$, and $R = a_2 h^{b_2}$, where a_1 , a_2 , b_1 , and b_2 are constants for a given cross section. Also in Eq. 2, $\gamma = 2b_2/(2b_2 + 3b_1)$ when using Manning's equation. For overland flow and flow in wide channels, a_2 , b_1 , and b_2 equal one and $\gamma = 0.4$. Details for solving Eq. 2 in a discretized raster model are discussed in Saghafian and Julien (2).

If the excess rainfall intensity i is uniformly distributed in space, then $Q_e = iA$, and the term $1/i$ can be taken out of the integral. Then, the travel time is proportional to $i^{-\gamma}$:

$$T_t = Ci^{-\gamma} \quad (4)$$

where C is a watershed hydrogeomorphic index conglomerating spatially distributed characteristics such as surface roughness, slope, flow length, flow accumulation, drainage pattern, and channel geometry. For the special case of watersheds with wide channels, γ equals 0.4.

Consider a simple case of an overland plane of length L and slope S_0 . Equation 2 reduces to the following equation

for travel time from the upstream of the plane ($X_1 = 0$) to its lower edge ($X_2 = L$):

$$T_t = \frac{n^{0.6} L^{0.6}}{i^{0.4} S_0^{0.3}} \quad (5)$$

The travel time obtained from Eq. 5 is essentially the time of concentration of the overland plane. For travel time in channels, Eq. 2 can be applied similarly with knowledge of the required parameters.

In practice, however, the application of these derivations is time-consuming and requires data that may not be readily available. In the following section, we describe the most commonly used T_c methods. Comparison of the such methods with the theoretical derivation can help to evaluate practice results versus theory.

METHODS AND FORMULAS

Numerous references have discussed different methods and formulas to compute T_c . However, some hydrology literature is not clear in the definition of appropriate terms in formulas, applicability, limitations, condition of original data used in developing the formulas, and even the units. In some methods, variables such as length, slope, and roughness are used with slightly different meanings. It is not possible to review all methods here, hence the discussion is limited to a few more widely practiced methods described in well-known hydrology references.

There are two general approaches for estimating T_c : hydraulic-based and empirical. The hydraulic laws of free surface flow are applied in the former approach in which the travel times for sequential reaches are added up. In the latter approach, however, empirical formulas are derived from field data, and the watershed is treated mostly as a lumped unit. One example of hydraulic-based methods is the method outlined in the previous section. This section proceeds with description of other hydraulic methods.

Hydraulic Methods

The kinematic wave model is the preferred method for the overland flow portion:

$$T_c = 0.94n^{0.6}L^{0.6}i^{-0.4}S^{-0.3} \quad (6)$$

where T_c is in minutes, n is the Manning roughness coefficient, L is the flow length (in feet), i is the rainfall intensity (in inches per hour), and S is the slope (ft/ft). The coefficient of Eq. 6 becomes 0.88 in SI units with T_c in seconds, L in meters, and i in meters per second. In theory, as in Eq. 5, the coefficient must be one. The resulting difference may be partly attributed to flow convergence. Typical values of overland Manning's n are given by Engman (3), Ponce (4), and McCuen (5). It is recommended that L not exceed 300 feet. Iteration is required in applying Equation 6 for the rational method because both T_c and i are unknown but related through intensity-duration frequency (IDF) curves.

The Soil Conservation Service (6) proposed the average-velocity method for T_c , which can also be applied more broadly to compute T_t , as follows:

$$T_c = \sum_{j=1}^N \frac{L_j}{60V_j} \quad (7)$$

where T_c is in minutes, L_j is the length of the j th flow segment (in feet), V_j is the velocity through the j th segment (in feet per second), and N is the number of segments. This equation and a similar one proposed by Chow et al. (7), clearly follow definition 1 that is based on flow velocity rather than wave velocity. The hydraulically longest flow path can be divided into N segments according to different flow types: sheet, concentrated, or channel. The segments should be approximately uniform in slope, roughness, and cross section (for channels). Charts of overland flow velocity are provided as a function of slope and cover by the SCS (6). The charts are part of a method known as the "upland method." A curve that represents concentrated flow velocity in small upland gullies is also provided (6). The flow velocities generally vary from 0.2 ft/s for mildly sloped forest lands to 15 ft/s for steep paved areas.

Equation 7 may be used to estimate the travel time through a channel. Manning's formula corresponding to a 2-year or bankful discharge may be used to compute the average velocity. For greater design return periods and/or higher than bankful discharges, however, average flow velocity must be computed considering the floodplain condition. In using Eq. 7, note that wave celerity may be 1.4 to 2.0 times the water velocity (1). So there are relatively large errors in computing channel travel time based on average water velocity. Therefore, using the wave celerity in computing travel time is recommended. Theoretically, the kinematic wave celerity is equal to βV in wide channels, where $\beta = 1.67$ for Manning's equation and $\beta = 1.5$ for Chezy's formula (4). If the Muskingum storage time constant (K) is already determined for a reach, this value can be used as a reasonable measure of wave travel time through that reach.

Linsley et al. (1) suggested station hydrograph analysis to obtain travel time in a stream. If the local inflow between the stations is negligible, travel time can be determined from the lag time between the hydrograph peaks measured at the two stations. Travel time is not a constant characteristic for a reach, so a range of stage-discharges is necessary to study the change in travel time as a function of stage. Travel time is expected to decrease as stage increases to bankful, and then it increases at stages higher than bankful.

Empirical Methods

Most empirical methods developed to estimate time of concentration assume the watershed as a simple lumped unit. Kirpich (8) presented one of the most commonly used formulas for estimating T_c based on data from some small watersheds in the United States. For Tennessee watersheds ranging in size from 1 to 112 acres, slopes from 3% to 10%, and well-defined channels, the following

formula was proposed by Kirpich (8):

$$T_c = 0.0078L^{0.77}S^{-0.385} \quad (8)$$

where T_c is in minutes, L is the length from the farthest point along the channels to the outlet (in feet), and S is the slope along L or approximately the average watershed slope (in feet per foot). T_c is multiplied by 0.4 for concrete and asphalt overland surfaces, and by 0.2 for concrete channels. With units of L in km and S in m/m, the coefficient in Eq. 8 becomes 3.97.

The Bransby-Williams formula (9), which is used for watersheds up to 50 sq. miles in size, is written as

$$T_c = LA^{0.4}D^{-1}S^{-0.2} \quad (9)$$

where T_c is in hours, L is the longest distance from the watershed boundary to the outlet (in miles), A is the watershed area (in sq. miles), D (in miles) is the diameter of a circle with area A , and S is the average elevation drop along L (in feet per 100 feet). With units of T_c in minutes, L in km, A in km², and S in m/m, and by converting D to A :

$$T_c = 14.6LA^{-0.1}S^{-0.2} \quad (10)$$

Equation 10 is one of the most widely used formulas.

A variant of the Kirpich formula was developed by the USBR (10) for small mountainous watersheds in California:

$$T_c = 60(11.9L^3/H)^{0.385} \quad (11)$$

where T_c is in minutes, L is the length of the longest stream (in miles), and H is the elevation difference between the watershed divide and the outlet (in feet).

Johnston and Cross (11) presented a relationship for T_c derived from the data of several watersheds ranging in size from 25 to 1624 sq. miles:

$$T_c = 4.7r^{-2}(L/S)^{0.5} \quad (12)$$

where T_c is in hours, r is the stream branching factor, L is the length of the main stream (in miles), and S is the average stream slope (in feet per mile). One may alternatively omit r and use 5.0 as the coefficient of the equation.

The SCS lag equation, developed from data on agricultural watersheds, can be modified by assuming that T_c is equal to 1.67 times the watershed lag:

$$T_c = \frac{1.67L^{0.8}[(1000/CN) - 9]^{0.7}}{1900S^{0.5}} \quad (13)$$

where T_c is in minutes, L is the length of the longest flow path (in feet), CN is the SCS curve number, and S is the average watershed slope (in %). This equation has been found acceptable for urban watersheds smaller than 2000 acres in size with completely paved areas.

GENERAL RECOMMENDATIONS

Selection of the equation for estimating T_c and T_t is subjective. Some general recommendations, however, can

be proposed. First try to make up your mind about the proper definition of time of concentration and travel time. Definition 2 is based on wave travel time and is preferred, but water velocity in definition 1 can be also transformed into wave celerity. Use definition 4 when hyetograph-hydrograph data are available, although rainfall spatial variability in large watersheds and nonreproducibility of response time are two major issues that are not addressed in this definition. Another important consideration is the preference to estimate T_c and T_t as functions of design rainfall intensity in an iterative procedure.

Next is the choice between hydraulic and empirical methods. Choose a hydraulic method when proper data can be collected to apply it and the estimate of T_c is considered a critical step in the project. If repetitive use of the selected method for various watersheds and for many outlet points inside the watershed of interest is required, the development of a GIS-based computer model based on Eq. 2 may be justified. Curves of T_c can be constructed for any watershed by running the model for various rainfall intensities. This leads to calibration of γ and C in Eq. 4.

The next option in hydraulic-based methods may be the combination of the kinematic wave model (Eq. 6) for overland flow, the upland method for concentrated flow, and Manning's equation for channel flow. Travel time through channel reaches must be adjusted based on wave celerity, as discussed in "Hydraulic methods."

In using empirical methods, one must be aware of the conditions and assumptions on which the development of these methods was originally based. It is important to compare the characteristics of the watershed considered with the range of watershed area, land use/cover, slope, and rainfall intensity of the original data set used in developing that empirical method. Simplicity and practicality of empirical methods comes at the expense of lost generality and accuracy.

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WATERSHED HYDROLOGY¹

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INTRODUCTION

Consideration of a drainage basin as the basic unit of supply and management goes beyond merely calling attention to the need for watershed management based on what we typically visualize of the natural environment. This article seeks to establish a broad base for anyone who wishes to base policy decisions on sound science, in this case, on some readily identified and easily comprehended fundamentals about natural and disturbed hydrologic environments. This article organizes, defines, and explains the importance of seven basic functions of watersheds, augmented by some essential topics aimed at a more thorough understanding of the functions.

It is essential to consider a watershed as a basic unit of landscape for managing the water resource. The reasoning underlying this practical concept is actually rooted in the fundamental role of water on this planet more than on the obvious practical management needs. The watershed is, first, a *system* whose primary focus is on interactions between the topography and runoff-producing events. Topography defines a watershed's limits and is a useful framework within which to examine, describe, and manipulate its *functions*, or how the watershed works. Runoff-producing events, rain, snow, or ice storms, and snowmelt induce stream flow that provides sites for chemical reactions and habitat for wildlife as well as pathways for energy, nutrients, and waste products between the land and the oceans. Second, the watershed is a dynamic *hydrologic unit*, whose behavior interacts with biological systems to construct a range of *aquatic ecosystems* that have unique properties reflecting the *quantity*, *regimen*, and *quality* of the water resource. These watershed functions blend on larger geographical and temporal scales with physical, chemical, and energy components of the universe, thereby affording the opportunity for humans to establish working landscapes that are environmentally sensitive, sensible, and sustainable.

Terminology: Three Important Terms

A watershed (also *catchment* or *drainage basin*) is the natural unit of land on which water collects, is stored, and

¹The material in this article is summarized from *Watershed Hydrology*, Second Edition, 1996, CRC Press/Lewis Publishers, Boca Raton, FL, and, in part, from an animated and narrated nutshell autotutorial short course on compact disk accompanied by a manual under the title *Concepts of Watershed Hydrology*, published by the author in 2002 (www.waterbudget.com).

is redistributed to the atmosphere, streams, and larger water bodies. The simplest definition is “an area of land on which precipitation collects and runs off through a common outlet.” The “outlet” is the lowest place on the watershed; it is selected as the best location for monitoring stream flow, its quality, and temporal distribution. The term also applies to the high ground between two drainage basins, which is also known as a *topographic divide*.

Hydrology is *the study of the movement and storage of water in the natural and disturbed environment*. The term also refers to the *condition of the hydrologic environment at some specified time or place*, as in “the hydrology of Eagle River just prior to the June flood. . . .”

Antecedent moisture conditions refers to *a description of meteorologic, vegetative cover and/or land use circumstances, soil characteristics, and the amount of water in storage on a watershed that collectively affect runoff behavior at the start of a hydrologic event of interest*.

Units

Both English and Metric System units are used throughout aquatic disciplines. Most of the well-established stream flow and water quality records made during the twentieth century in the United States and Great Britain were in units of inches, feet, and acres. Other regions of the developed world used the cgs (or metric) system of millimeters, meters, and hectares. Hydrologists need to be familiar with both sets of units and should be able to convert readily from one to the other. Each has benefits. For example, conversion from water quality units of *concentration* in milligrams per liter (mg/L) to *load* in kilograms per hectare is easier in the metric system. The large volume of water represented by the *acre-foot* (43,560 square feet covered by water one foot deep) can be readily derived from the basic unit of stream flow in the English system, one cubic foot per second (cfs, or second-foot), which is a reasonable minimum measurable flow in the field. Thus, a constant flow of 1 cfs for 1 day is very close to 2 *acre-feet/day*, or about a third of a million gallons.

WATERSHED FUNCTIONS

The seven watershed functions are arranged in three groups: three *hydrologic* functions, two *ecological* functions, and two *response* functions. Large watersheds are made up of many small watersheds, so the discussion here focuses on *small* watersheds, where stream flow is dominated by storm flow or discharge from runoff-causing events.² How runoff events from small watersheds combine to produce discharges on large watersheds is the subject of extensive research and complex models

²A small watershed has been defined as (1) from 200 acres to 1 square mile in size for applying the Rational Formula to determine peak flows (1); (2) one in which runoff behavior is dominated by *overland flow* as opposed to *channel flow* (2), and, for legislative/programmatic purposes; and (3) one that is up to 250,000 acres (PL 566, 1954). It is suggested here that the small watershed be defined *functionally* as one on which the water resource issue of concern be under the primary influence of *storm flow* as opposed to *baseflow*.

and research techniques. Runoff from large watersheds is dominated by baseflow (discharge from a groundwater reservoir). The behavior of these two types of flow is very different, but once the water is in the stream, visual separation is often difficult. In contrast, however, the chemical characteristics of storm flow and baseflow are also quite different: they have different storage histories that are reflected chemically. That provides a developing field of research into using water chemistry to understand, model, and therefore predict hydrologic behavior.

Hydrologic Functions

The names of the hydrologic functions derive simply from what they do. The British term for the drainage basin is *catchment*; the U. S. term is *watershed*. Both are self-descriptive. Between them, the water is *stored*.

Collection. The collection function is clearly the first step in a process-oriented description of the hydrologic cycle on land.³ An essential first consideration in this function is where, how, for how long, and in what form the runoff-causing event impacts the watershed. For example, the same rainstorm at different locations on a watershed will result in different distributions of stream flow (Fig. 1). Thus, a small-area but intense⁴ summer thunderstorm near the outlet of a watershed will produce a short-duration, high-peak runoff flow in contrast to the same storm occurring at the upper end of the watershed where the length of stream provides time for the runoff to spread out in time and space, thereby producing a lower peak but longer duration of flow. The hydrograph for the storm near the outlet is said to be “flashy,” a common characteristic of small watersheds as well as impervious parking lots and urban watersheds.

It is clear that the location of the runoff-producing event is of considerable importance in determining hydrologic behavior. Size and type of storm, therefore, also become important. Thus, in a region where summer thunderstorms are prevalent, short-duration, flashy stream flow may be expected, especially if the watershed under consideration is at a high elevation where slopes are steeper, soils are thinner, and intense storms can readily cover the entire small watershed. These are *weather* events. In contrast, snowmelt is a *climatic* event that embraces a longer time and greater areal extent than thunderstorms. In between, cyclonic storms and hurricanes may be the dominating determinant of runoff behavior on intermediate size and midelevation watersheds. Climate and weather are characteristic of different areas, so hydrologic behavior is similarly geographically determined.

Summarizing, the collection function reflects how and where the water that falls on a watershed influences the storage and discharge functions. Further, the different types of runoff-causing events interact with and define

³There is, of course, a hydrologic cycle on waterbodies, too. It consists largely of the same functions as those on terrestrial (land) environments, but in the oceans, without the discharge, attenuation, or flushing functions.

⁴Expressed in inches or millimeters per hour.

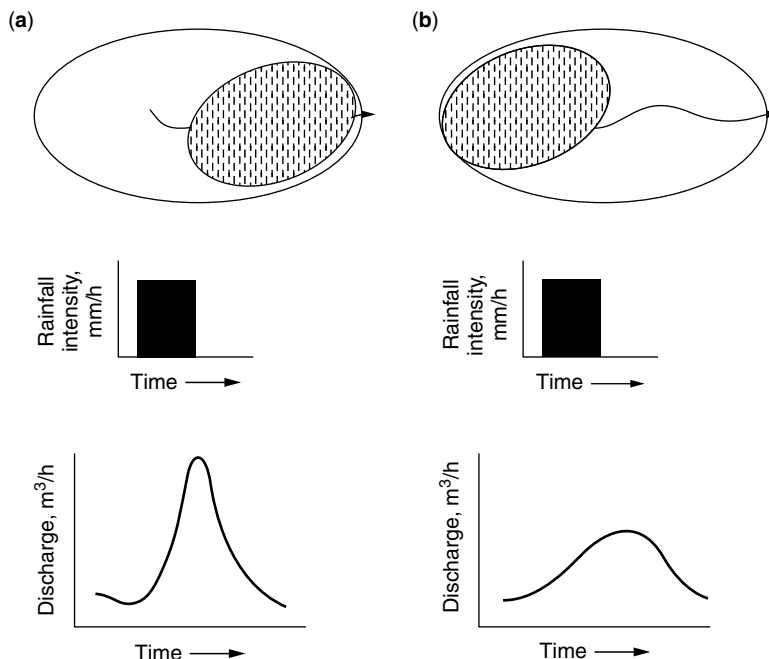


Figure 1. Storm hydrographs from the same rain-storm (hatched area) at different locations on the same watershed. (a) Storm is near the outlet. (b) Storm is near the headwaters. The volume of water running off might be the same (the area under the hydrographs), but the time pattern of runoff is different.

typical *hydrographic regions* in which homogeneous areas exhibit similar hydrologic behavior.

Storage. The storage function is the focus of a small watershed (Fig. 2). There are six types of storage in and on a watershed: *channel*, *depression*, *retention*, *detention*, *groundwater*, and *vegetation* storage.

When water arrives at the land surface, three things can happen to it. First, water may fall directly into channels, including small lakes and ponds, as well as streams and rivers. From these, of course, it is simply a continuing process for this water to flow out of the watershed by joining other waters in the stream and be reclassified as channel runoff.

Second, water may be held on the surface as *depression* storage in a thin film or as puddles. Wetted surfaces of all types—on blades of grass or leaves, rooftops, parking lots, even in some cases wetlands—may be considered depression storage. Snowpack is also a form of depression storage. The water thus detained on the surface may enter the soil if the pores in the soil are not already full and are large enough to allow water in; often, water in depression storage evaporates from the surface under the influence of wind, a warm surface, or sunlight following the storm. Runoff from depression storage is minimal.

Third, water may infiltrate (move into) the soil. How much and how fast this occurs depends on the openings or pores in the soil, whether there are small soil grains that can be washed into and clog the pores, how much of the soil pore space is already filled with water, and how much and how fast the water already in the soil can drain downward (*percolation*). Some larger scale factors that affect the infiltration rate⁵ include surface roughness, slope of the land, and extent of soil particle

size variety, often dependent on the geologic history of the area and its erosion (known collectively as *geomorphology*). If the rate at which the precipitation falls—*rainfall intensity*—is greater than the capacity of the soil to absorb it—*infiltration capacity*—then excess water runs off the top of the soil as *surface runoff*.

Once in the soil, water may be held tightly (close to soil surfaces) and for long periods in the capillary (or *micropores*) against the force of gravity, or it may be held temporarily (not as close to soil particle surfaces) in the larger, noncapillary or *macropores*. The water *retained* for a long time is called *retention storage*, whereas the water temporarily *detained* is *detention storage*.⁶ Upon infiltrating the soil, water will move first into retention storage if it is not filled (where the water is pulled in and held at higher tensions) and into detention storage only if the retention storage is already filled, or recharged.

Water in retention storage is held tightly against the force of gravity, so water cannot flow out of retention storage. In contrast, water can flow out of detention storage and is referred to as subsurface runoff.⁷ Subsurface flow may go either to a nearby stream or to a *groundwater* reservoir, the body of water beneath the *water table*. A groundwater reservoir is the saturated zone from which water tends to flow off rather slowly as *baseflow* in contrast to the greater degree of flashiness exhibited by *storm flow*. The latter is made up of channel, surface, and subsurface runoff, sometimes referred to as “wet weather flow.”

Topography (and geological history) plays a significant role in the development of soil storage reservoirs. Elevation, slope, aspect (of slopes), orientation (of the

⁵The rate at which water moves across the air–soil interface, in inches or millimeters per hour.

⁶The difference may be readily committed to memory by recalling the term used to describe the punishment for improper behavior in school when a student would be sent to a temporary hour-long study hall (*detention*), not commitment to life in prison (*retention*).

⁷The terms “runoff” and “flow” are often used interchangeably.

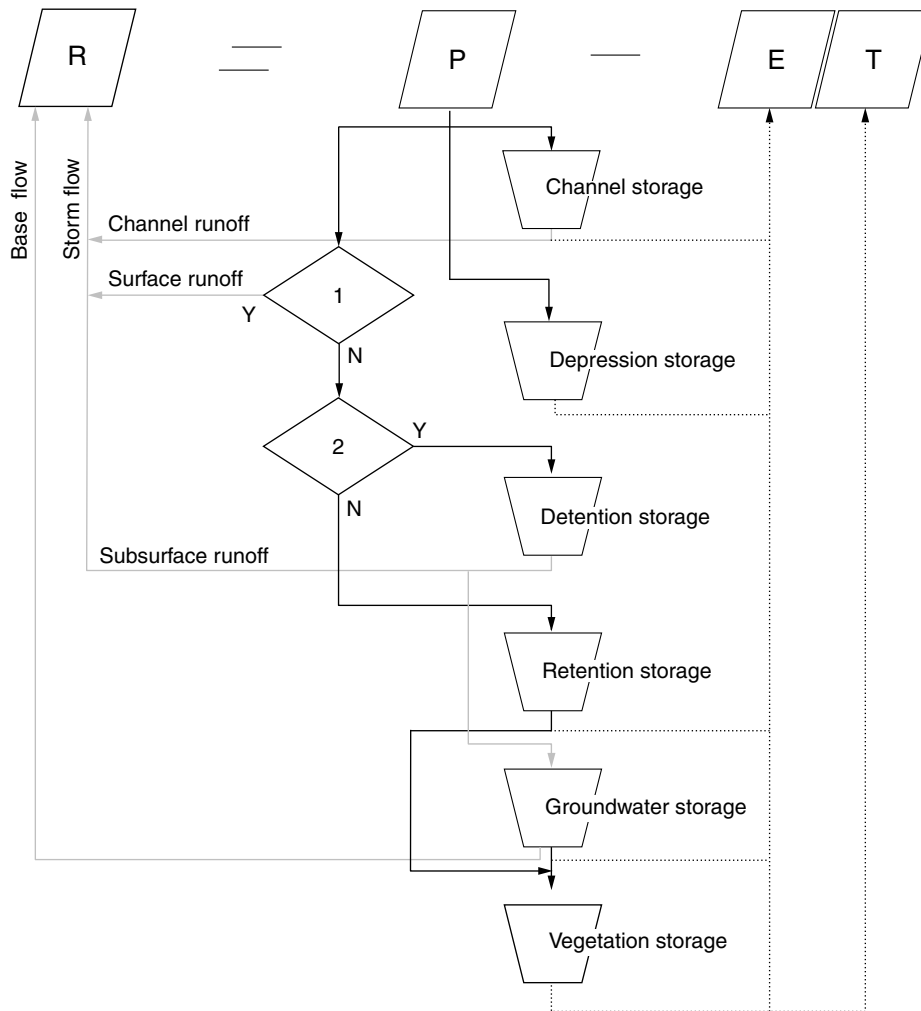


Figure 2. The hydrologic cycle depicted by focusing on the six types of watershed storage and the basic water balance equation ($R = \text{Runoff}$; $P = \text{Precipitation}$; $E = \text{Evaporation}$; and $T = \text{Transpiration}$) at the top. Choice box #1 asks “Does precipitation intensity exceed infiltration capacity?” and choice box #2 asks “Is retention storage full?” Some features, storm flow and baseflow, are kept separate to emphasize that they behave differently; water can leave retention storage only through evapotranspiration (E_t) processes; and water in detention storage, which flows out in response to gravity within about 24 hours, generally is not around long enough to supply E_t at all. See text for full description.

stream), and the distribution of those features throughout the watershed affect the antecedent moisture conditions as well as the storage characteristics of soil reservoirs. For example, south-facing slopes in the Northern Hemisphere are drier than their northern counterparts because they receive more incident energy that is used to evaporate moisture. Soils on south-facing slopes are less well developed and thinner, have lower water holding capacities, and a greater percentage of the soil pores in detention storage where less water is available for vegetation, therefore fewer and/or different species. In short, south-facing slopes tend to be drier than north-facing slopes in the Northern Hemisphere and, of course, the opposite south of the equator.

Finally, water may be stored in *vegetation*. Water enters vegetation when vapor condenses on plant tissues, when it is intercepted as rain or snow, or when absorbed directly through leaves and bark. More water enters through the roots, however, and, because it ascends in response to atmospheric evaporative tension working through tiny openings in leaves called stomates or stomata,⁸ it is often combined with the water that evaporates from depression (surface) storage on the vegetative parts as

evapotranspiration (E_t). E_t refers to all of the evaporative processes and may be modeled as an object of research that focuses on the amount of energy available for evaporation (3). Evapotranspiration is strong enough to pull water away from high-tension films close to soil particles (retention storage). In contrast, water in detention storage is held loosely in the macropores and flows out usually within 24 hours; during that time there usually are waning storm clouds, high humidity, and often cooler temperatures all of which diminish the energy (and water) available for evapotranspiration. The bottom line is that water held in detention storage is not generally available to plants or E_t .

From this examination of the storage function, it is clear that storage type and distribution in the soil profile and topographically over the watershed modify the broad hydrographic patterns imprinted by the climate and weather factors that characterize the collection function. The infiltration capacity concept and the distribution of retention and detention storage are two of the most important concepts in watershed hydrology. And, in practice, the relationship between infiltration capacity and precipitation intensity combines to define the limits to surface runoff and attendant flooding that occurs at the

⁸The process is known as *transpiration*.

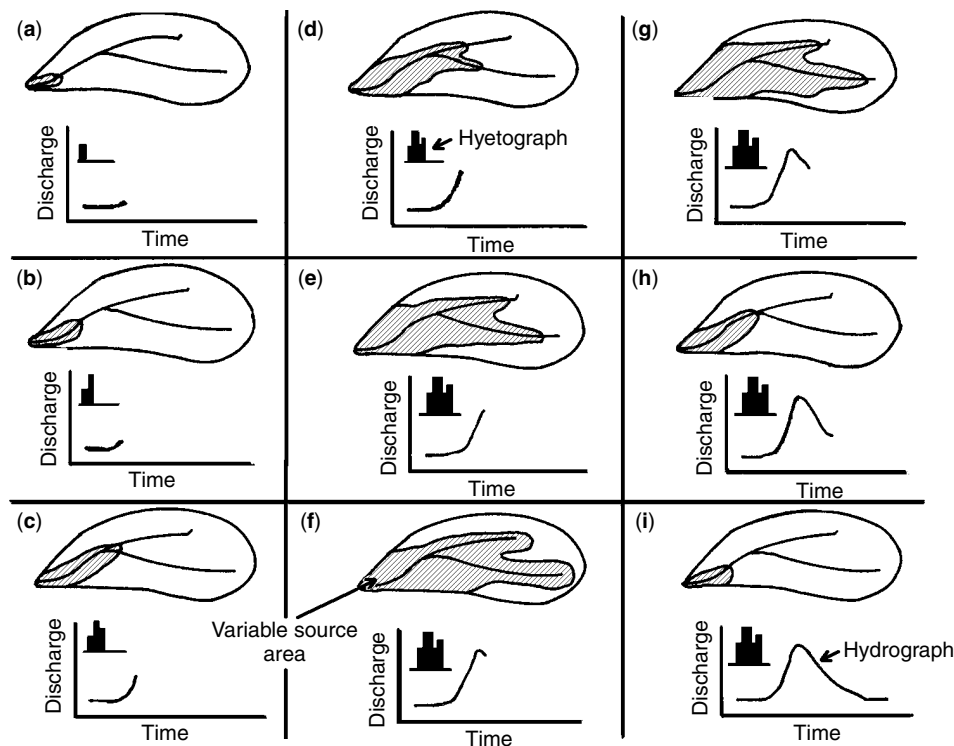


Figure 3. The variable source area. During a storm that uniformly, instantaneously, and completely covers a watershed, the zone contributing to runoff changes.

air–soil interface where humans have the greatest impact on the hydrologic cycle.

Summarizing, the storage function focuses on how water is retained and detained on a watershed and what its characteristics are as it moves between storage reservoirs. Storm flow dominates small watershed runoff behavior. The fact that water can readily absorb elements and compounds gives rise to the chemical function (below) and makes it possible to use water chemistry to monitor and model water storage.

Discharge. As we have just seen, there are two categories of runoff: *baseflow* and *storm flow*, which include channel runoff, surface runoff,⁹ and subsurface runoff. Storm runoff is ramified in the *storm hydrograph*, a plot of runoff over time that translates unique watershed characteristics into a typical discharge (often, a flood). Changing land use or vegetative cover can profoundly affect the storm hydrograph on small watersheds and, if done over a wide enough area, even on large ones. The baseflow hydrograph also exhibits patterns that are unique to the watershed from which it flows. However, baseflow integrates the runoff over longer times and broader landscapes, so it is less sensitive to land use and vegetative cover changes. Both types of flow display typical patterns for hydrographic regions, allowing extrapolation of hydrologic information from one watershed to another within the same region. Thus, regional flood behavior curves may be estimated for ungauged streams. The

storm hydrograph is influenced primarily by *weather*; the baseflow hydrograph is influenced primarily by *climate*.

An important concept of discharge is that the portion of the watershed that contributes to storm runoff changes during the runoff-causing event, even if uniform rainfall that instantaneously and completely covers the watershed is assumed. The variable source area model identified by Hewlett and Hibbert (4) promotes understanding how and when water gets to the stream (Fig. 3). It also provides a sound scientific basis for protecting and restoring the near-stream *riparian zone* areas, also referred to as *buffer zones* or simply as *buffers*. Water flowing off or through the soil, or in saturated stream bottom zones, appears rather quickly as storm flow, so it follows that water from these zones is likely to be a major source of nonpoint sources of pollution.¹⁰ *Isotope hydrology*, which monitors specified elements or ions in the environment, is one method of tracking water in time and space as it moves through natural and disturbed environments enabling identification of how “old” and “new” water contributes to stream flow. Other isotopic elements and dyes may also be used.

Summarizing, the discharge function integrates antecedent moisture conditions and characteristics of the collection and storage functions. Storm and base flow portions of runoff need to be analyzed separately because they arrive at the stream from different watershed storage reservoirs, are influenced by different characteristics, and consequently behave differently. Both the *storm* and *baseflow* hydrographs are unique to a particular watershed. Hydrographs may be used to analyze, describe, and

⁹Under undisturbed and most natural conditions, surface runoff (also referred to as *overland flow*) does not occur. Exceptions are evident on steep slopes that have shallow soils and intense precipitation.

¹⁰Pollutants that are diffused across the surface of the land and have no apparent end-of-the-pipe outlet as do point sources.

predict stream behavior on ungauged watersheds. A great deal may be learned about the watershed by careful analyses of the storm and baseflow hydrographs and their relationships to other hydrologic parameters (e.g., precipitation, infiltration, and soil storage) of the drainage basin.

Ecological Functions

Ecological functions refer to *chemical reactions* and *habitat*. Except for nuclear reactions, virtually all chemical reactions on the earth, in our solar system, and throughout the universe, take place in the presence of water, which therefore plays an essential role in environmental chemistry and life itself. The aquatic chemistry of our body cells, organs, and blood, in soil water, streams, wetlands, ponds, and lakes, and in the oceans define life. Life is considered as existing only in the presence of water and, consequently, the ongoing search for extraterrestrial life focuses on a search for water.

Chemical Reactions. As drops of water form in the atmosphere following cooling and condensation, carbon dioxide readily dissolves in them. This shifts the *reaction*¹¹ (or pH) of natural rainfall from a neutral value of 7.0 to about 5.7 (slightly acid), which makes precipitation rather corrosive; it dissolves many of the elements and/or compounds found in rocks and soils that derive from them. Carbon dioxide, dihydrogen oxide (water), and several forms of carbonate all disassociate (to hydrogen and hydroxyl ions) in aquatic environments and result in various concentrations of hydrogen ions; hence the use of pH to monitor one of the major water quality parameters. Others include *conductivity*, an index to dissolved salts based on how much electricity the water conducts; *temperature*, a measure of the energy content of the water; *turbidity*, a measure of the amount of suspended solids in the water; *dissolved gases* (oxygen, carbon dioxide, methane, ozone, etc.); *dissolved nutrients* (nitrogen, phosphorous, potassium, and sulfur); *heavy metals* (iron, lead, mercury, etc.); *color*, an indication of dissolved substances such as tannin; and *contaminants*, including toxins and disease-causing bacteria and/or viruses, and spores.

Some of the chemical exchanges that are evident in the presence of water may, in turn, be used to assist in evaluating the hydrology of a waterbody: where it came from, with what it has come in contact, and so on. For example, silica dissolves readily when water moves through sandy soil. If that water flows into a wetland, abundant plant life will remove silica from the water column to produce cells, which differ from animal cells in that they have silica-based cell walls. And, because the great growth of plants in wetlands ensures a plentiful supply of decayed plant and animal remains, water flowing out of a wetland will tend to be low in oxygen and high in carbon dioxide (as well as low in silica and pH).

¹¹A measure of the relative acidity and alkalinity of a compound in terms of the hydrogen ion concentration. The range is from 1 to 14 (the numbers are logarithms to condense an otherwise very broad scale), where 7.0 is neutral.

Water is considered a universal solvent (although it won't dissolve the noble metals gold, silver, platinum, etc.) and naturally conveys waste materials from our body cells, organs, individuals, communities, and nations. It even assists in carrying nutrients *upstream* in the bodies of migratory salmon and related species that feed in the oceans and die when they have completed their reproductive journey to the headwaters of the stream of their birth, ironically, succeeding because of the unique stream chemistry that they can trace to the stream in which they were spawned. It is also believed that it plays an essential role in the carbon cycle (5).

The hydrologic cycle also interacts with the energy balance of the earth in two ways. First, water vapor is one of the chief greenhouse gases along with carbon dioxide, methane, ozone, and other two-atom molecule gases. Second, water vapor in the form of cloud cover plays a direct role in the energy balance of the planet by reflecting solar radiation to space, thereby affecting the temperature of the atmosphere and the earth's surface. We need to explore ways in which our land and water management strategies and policies might adversely or beneficially impact the role of water in these critical interactions.

Summarizing, the *chemical reaction* function brings a whole new and exciting dimension to the field of watershed hydrology and land management, not only water chemistry, but environmental and astrophysical chemistry. These are some very significant and fundamental characteristics of the earth and the universe.

Habitat. The habitat function embraces the observation that wildlife (terrestrial, aquatic, and avian) evolved under a variety of conditions and require that they be maintained to survive. Ecosystem support for all species needs to be preserved and/or restored to ensure continued biodiversity in our environment, especially aquatic environments. Human beings themselves may have evolved at the beginnings of civilization at the seashore, in the presence of water and all that it has to offer as sustenance, food, protection, and a myriad of services (6).

This function is not dealt with in depth in this paper, but it is important to point out that a great deal of time and money have been invested in restoring (and preserving) wetlands, and recently increased activity in watershed restoration as well. These preservation and restoration efforts are often lackadaisically directed toward restoring habitat rather than hydrologic functions. But, when habitat—particularly aquatic habitat—is restored, the chances are very good that hydrologic functions have been restored as well.

Summarizing, the habitat function might be viewed as a surrogate for evaluation of restoration of watershed functions that are harder to visualize, measure, or model. The importance of viable ecosystems and associated biodiversity, however, is a value of aquatic habitats that transcends even semiserious personal desires.

Response Functions

The response functions are so called because they represent practical functions that play important roles

in our aquatic environments. The responses really involve how the storm hydrograph reacts with other considerations: in the *attenuation function*, the peak discharge, Q_p , is seen to diminish as the area of a watershed increases. In the *flushing function*, the concentration of dissolved and/or suspended substances appears to relate to the stages of the storm hydrograph in a predictable manner.

Attenuation. It is the nature of storms to have the most (depth) and most intense (depth per unit time, e.g., inches per hour) precipitation at or near the center, and precipitation diminishes until both become zero. This observation flies in the face of the earlier assumptions of uniform, instantaneous, and complete storm coverage: in fact, the larger the watershed, the longer it takes for water falling at any point to reach the outlet, in addition to the fact that there will be a place where the rainfall is most intense and greatest in depth. Given, too, that slopes and stream gradients generally diminish downstream, it follows that the length of time between the start (or centroid) of the runoff-causing event and the time of occurrence of the consequent peak flow Q_p (in this case, Q_p is reported in units of cubic measure per unit time *per unit area*, e.g., cubic feet per second per square mile, csm, or cubic meters per second per square kilometer) also diminishes (Fig. 4).

Summarizing the attenuation function, it is easy to remember that the three "Ls" go together: thus the storm peak in runoff per unit area is *Lower* and *Later* on *Larger* watersheds.

Flushing. Flushing is a natural characteristic of all aquatic ecosystems except the ocean. As relatively pure water arrives at the land surface (or channels), it dilutes whatever substances are already in the stream (Fig. 5). Then, as water flows off the near-stream zones within the variable source area, it carries with it soil particles and dissolved materials and starts a cleansing or flushing action in the stream/watershed system. As the velocity and volume of stream flow increases, more and more particles deposited earlier in the streambed may also be picked up and transported downstream. A plot (*pollutograph*) of the concentration (e.g., mg/L) of dissolved and suspended

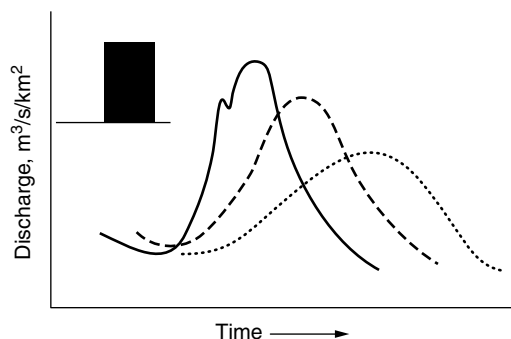


Figure 4. The initial upstream storm hydrograph diminishes and spreads out as the peak moves downstream. Note: the vertical axis is in volume *per unit area*, and the initial channel interception prepeak on the rising limb disappears.

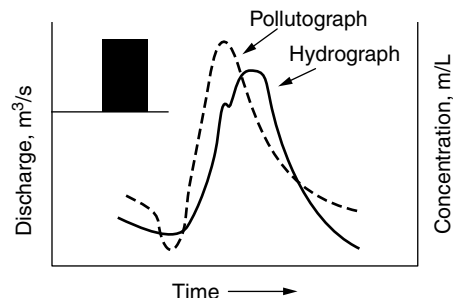


Figure 5. Flushing of some suspended solid (read on right-hand axis) superimposed on storm hydrograph (read on left-hand axis).

materials increases until peak flow occurs and then quickly diminishes. Put another way, the plot of the concentration over time, the pollutograph, peaks during the rising limb of the storm hydrograph (or during the rising stage of the annual hydrograph's spring peak, for example), then rapidly decreases until the prerunoff-causing event stream flow level is reached.

Some of the potential pollutographs are well known. For example, a great deal of field data and physical modeling have produced reliable suspended-sediment pollutographs. However, there is a paucity of data and studies on pollutographs for dissolved gases, pH, temperature, and dissolved solids. Intuitively, pollutographs generally follow this pattern. But there are not enough studies on this topic to back up that assertion.

Summarizing the flushing function, nearly pure water from runoff-causing events dilutes water in the stream and initiates dilution and then a flushing action, within and near the stream, that peaks during the rising limb and returns to the preevent level commensurate with stream flow.

SUMMARY

Essential to life as we know it and expect that it evolved on other planets, water pervades our local, regional, national, and worldly environments. We have the greatest access to it locally, but our attention to careful science-based management of water on all scales is demanded to ensure our sustainability. Understanding water in the natural and disturbed environments starts with honoring basic hydrology on a watershed basis.

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WATERSHED MANAGEMENT FOR ENVIRONMENTAL QUALITY AND FOOD SECURITY

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Watershed management is a broad term. It comprises the planning, implementation, and operation of programs and practices for the optimal utilization and preservation of natural resources such as water, forest, and wildlife. The term is rooted in forestry, and the adverse impacts of deforestation relating to soil erosion, sedimentation, and on water yield are well known (1), and they need not be elaborated here. Suffice it to say that the frequency and severity of floods and droughts, mudslides, and water quality are closely linked to the state of watershed management. In this article, however, watershed management is considered with specific reference to environmental quality and food security.

Environmental quality is an illusive term that embraces physical, psychological, and emotional factors, and biodiversity is an important aspect of the same. Furthermore, an increasing awareness of the critical interdependence of ecology and hydrology has led to the emergence of a new study field of ecohydrology (2). It is a frontier full of challenging and unexplored questions, embracing problems, which are crucially related to biodiversity and ecosystem functioning; e.g., how do hydrologic systems control (or how they are controlled by) ecological systems?

In a given landscape, its flora and fauna dynamically interact with the hydrologic cycle over varying spatial and temporal scales, and the ecological structure and function of ecosystems are intimately tied to the hydrological regime of floods and droughts. Accordingly, ecological diversity is a function of varying flow. Moreover, it is essential to not only identify these interactions, but also to ascertain the limits to perturbation before the ecosystems begin to break down. In this context, estimation of instream flows plays a vital role for the preservation of environmental quality (3). Furthermore, two other aspects are important as well. First, within an overall watershed management strategy, ecotourism development should be mild to moderate lest it endangers the very environment that the tourists want to experience. Second, it should be recognized that the places where biodiversity is greatest are often the same places where conservation measures are most difficult to enforce because of poverty, difficulty in controlling poaching, and habitat destruction.

Food security can be simply defined as the ability of a nation to feed itself by means of its agricultural resources, and such ability is largely dependent on irrigation and drainage infrastructure. Inadequacy of such infrastructure leads to poor crop yields and loss of fertile land because of salinization. However, central to irrigation is the availability of water resources, and it is the watersheds that generate this vital resource. Unfortunately, in some parts of the world, such as in the Middle East and the Caribbean (4,5), watershed management does not get the attention it deserves, because, virtual water, embedded in grain imports, creates an illusory sense of food security, which in turn slows the pace of socioeconomic reform needed to improve watershed management.

In conclusion, although watershed management policies are shaped in the political arena, it is hydrology that can steer decisions in the right direction. Therefore, political and economic pressures should not prevail over hydrologic evidence and reasoning. Ultimately, there is either science and sustainability or crisis and conflict in the sphere of watershed management.

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WATER HYACINTH—THE WORLD'S MOST PROBLEMATIC WEED

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INTRODUCTION

Water hyacinth (*Eichhornia crassipes*), a perennial, herbaceous monocotyledon member of the pickerelweed family (Pontederiaceae), is native to tropical America. It is usually found as a floating plant with mature specimens having long feathery roots that can be over 2 m in length (Fig. 1). However, in shallow waters, individuals can root themselves into sediment, and individual plants can even be found fully rooted on banksides. Water hyacinth is found in a number of differing morphological growth forms, and the morphology and architecture of leaves, petioles, and clonal groups can be highly plastic (1–3). However, two forms dominate (4,5). One, typically found in open

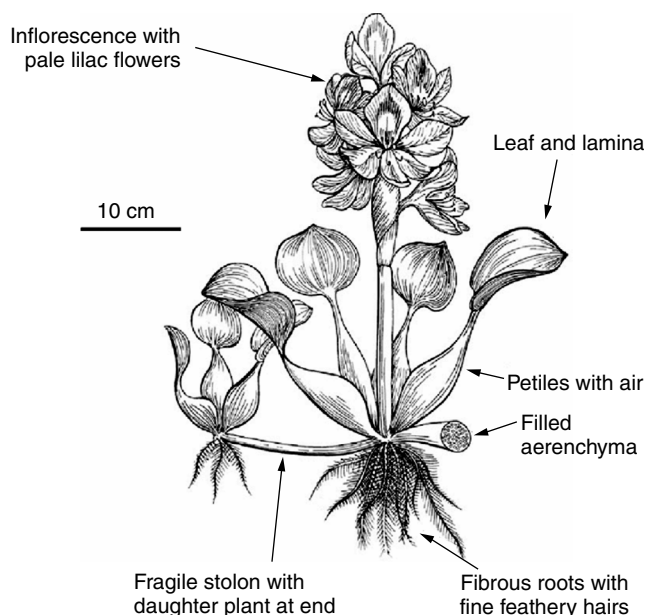


Figure 1. Water hyacinth (*Eichhornia crassipes*).

water or at the edges of plant mats, is characterized by short bulbous and very buoyant petioles containing air-filled aerenchyma, whereas another form occurs in crowded areas, such as in the middle of plant mats, with slender, tall, nonbulbous petioles. The short bulbous form can develop into the tall nonbulbous form if crowding occurs and sufficient nutrients are available (1). Petioles can therefore grow to over a meter in length, although 50 cm is usually more typical. Center and Spencer (2) found that in mature plants, between 6 and 8 leaves or lamina are present on each plant and that an equilibrium establishes between new leaves appearing and old leaves senescing.

Plants can double their biomass in 6 days (6) and although water hyacinth produces seeds, which form part of the waterbodies seed bank, plants generally reproduce through clonal propagation with daughter plants being produced at the ends of fragile runners or stolons. These stolons are easily broken by wind, current, or animal and boat movements, thus allowing the detached clones to disperse and colonize new areas.

In the center of each mature rosette of leaves rises a broad stem, on top of which is an inflorescence that usually bears between 6 and 14 lily-like pale lilac or violet flowers. Each flower consists of 6 petals, the upper one marked with blue and yellow, 6 stamens, and a 3 segmented ovary. The fruit that develops comprises of a simple capsule containing in the region of 400 to 500 seeds. After flowering, the inflorescence bends down into the water and seeds sink to the bottom of the waterbody where they can remain dormant for 20+ years (7) before germinating if the correct conditions prevail. For instance, a waterbody may become temporarily dry, resulting in all the parent plants dying; however, following the replenishment of water to the system, the seeds can germinate, thus providing a new stock of plants.

The fine feathery hairs that surround the main fibrous roots have a purple tinge and capture small particles as well as removing nutrients from the water column. Ideal growth conditions include a high nutrient concentration, especially of nitrogen and phosphorus, with water hyacinth thriving especially well where anthropogenic influences have artificially raised these levels. In addition, tropical temperatures, plenty of sunlight, and a high relative humidity all benefit the growth and expansion of water hyacinth.

With different morphological forms and an ability to adapt the growth form to the prevailing conditions, water hyacinth is able to colonize new areas very quickly. Although mats often develop out from the shoreline of lakes, rivers, and reservoirs, detached floating mats can also exist that are many kilometres in diameter and several meters thick. However, in its native habitat, the plant is kept in check through both natural pests and, more importantly, annual floods. During these periods, rivers may rise by over 10 meters, resulting in a torrent of water that sweeps the plants out to sea where they perish because of the effects of salinity. A few individuals remain, however, snagged on the banks and protected in quieter backwaters, which act as the new source of the following years stock. Overall, therefore, water hyacinth only becomes a significant problem when it finds its way into areas outside of its natural range.

THE PROBLEM

Water hyacinth was seen as a botanical speciality and, because of its beautiful and delicate inflorescence, used to decorate ornamental ponds (6,8). Therefore, from its native range, it was spread around the world during the nineteenth century. It was recorded in Egypt in the 1870s, the United States in the 1880s, Australia and southern Asia by the 1890s, China and the Pacific by the early 1900s, South Africa in 1910, East Africa by the 1930s, and West Africa by the 1970s. It is now established throughout tropical and warm temperate regions of the world (5,9). Moreover, as demonstrated in the Democratic Republic of Congo, where two plants were seen to produce 1200 daughter plants over four months (10), water hyacinth has massive growth potential. In terms of biomass, a m² of the weed can increase by up to 500 g each day so within a year, a hectare of plant can increase by over 1800 tons fresh weight, which, of course, does not include the subsequent growth of the new material. This exceptional growth rate is often partly attributable to the presence of large quantities of available nitrogen and phosphorus, and unfortunately, because anthropogenic eutrophication has affected the great majority of waterbodies throughout the world, the growth of water hyacinth and the problems associated with it have been greatly exacerbated. As such, and further aided by its ability to colonize new habitats and a lack of natural controls outside of its native range, water hyacinth is now commonly described as the world's most problematic aquatic weed (Fig. 2).

Invasive species such as water hyacinth have been referred to as a form of biological pollution because they can upset equilibria between native species that have



Figure 2. Lake Victoria in 1995. A mat of water hyacinth stretches out into the lake resulting in enormous ecological, sociological, and economic impacts.

formed slowly over millennia. When an invasive plant such as water hyacinth enters a new habitat it will out-compete and thus displace native vegetation, thereby becoming established. Some of the ways such displacements affect the native biota include a reduction in biodiversity, a decrease in habitat and food, and an overall change to ecological processes (11).

In the case of water hyacinth, native plants along the fringes of rivers and lakes are eliminated through successional shifts and, generally, a loss of biodiversity occurs (12). Furthermore, as native flora and fauna have coevolved over long periods of time, replacement of native vegetative systems with ones dominated by water hyacinth typically alters these relationships with accompanying reductions in available habitat and food. For instance, under the dense mats that can form, a reduced light and oxygen climate can lead to a decrease in plankton primary production, resulting ultimately in the death of native fish.

Ecosystem processes may change as a result of the spread of water hyacinth. Native ecosystems have developed under and adapted to particular abiotic factors and processes, such as rates of nutrient cycling, rainfall patterns, and fire regimes. The presence of alien species can alter these processes, which has a knock on effect to the ecosystem as a whole (13). For instance, where water hyacinth has been introduced, it can disrupt the natural hydrologic cycle by transpiring greater quantities of water, up to three times more, than native vegetation in the same habitat (11). Water tables and some surface water habitats are thus reduced and native species impacted.

Water hyacinth, with its characteristic thick fringes and floating mats, also has more direct impacts on anthropogenic activities. The weed can impede navigation, even in motorized boats, and transportation costs may increase because of the necessity of taking alternative

indirect routes or as a result of motoring through the mats, which, if physically possible, increases fuel consumption (14,15). The weed has affected fishermen by fouling their nets and traps, reducing catches, and preventing access to landing sites. In West Africa, the inland waters of Ghana, Burkina Faso, and Togo have been particularly hard hit. Here, not only has the weed slowed down the boats of fishermen, it has also made landing difficult to the point that more of their catch goes off (14). Fish are often an important source of protein to lakeside communities, and without it, health suffers. Moreover, the mats are breeding grounds for carriers of human and animal disease such as malaria, schistosomiasis, encephalitis, and river blindness (11,16). Irrigation schemes have been put under pressure and, indeed, irrigation channels blocked. In many cases, the weed has reduced the supply of clean potable water and caused difficulties in water extraction. Unsurprisingly, its presence has increased disputes between local communities and overall has been responsible for the translocation of some communities away from affected areas when local watering points and lake access have become unavailable. Furthermore, in Lake Victoria, apart from the reduction in national revenue from the reduced export of fish produce, tourism has declined and intermittent closure of the hydroelectric plant in Jinja, because of weed buildup, has caused disruption within Uganda's capital, Kampala (6,16,17).

Unsurprisingly, therefore, the World Bank, together with the United Nations, the International Monetary Fund, and a plethora of other international and national organizations, has pumped hundreds of billions of dollars into tackling the problem of water hyacinth around the world.

METHODS OF CONTROL

Once established, water hyacinth is extremely difficult, if not impossible, to eliminate, and attempts to do so often adversely impact remnant native vegetation (18). More achievable, however, is the control of the weed, but the elimination or even control of waterweeds is a costly process both in fiscal and temporal terms. Three basic methods exist to remove water hyacinth: physical, chemical, and biological controls.

Physical control involves employing manual labor and mechanical harvesters to remove the weed. The use of physical barriers, such as floating booms, to corral water hyacinth and prevent its spread as well as aiding ease of physical removal can also be implemented (14). Often, manual removal at landing stages can be undertaken extensively by locals if they are provided with equipment such as rakes and wheelbarrows. Aquatic vegetation cutters or "Swamp devils" have been used widely throughout the world, and booms are often used in conjunction to reduce weed expansion (19). These techniques have often worked in that weed can be contained and removed. However, often as a result of the sheer scale of the problem, benefits are typically short lived as a cleared site can be inundated with new weed in a matter of days, either through growth of any remaining

weed or immigration of plants from outside of the site. Furthermore, booms can become overburdened from the weight of the weed (20) and machinery can breakdown. Unfortunately, with a lack of expertise, tools, and spare parts, the expensive harvesters have, on occasion, been left to rust on the shore. However, physical control has been used successfully throughout the world to manage water hyacinth (21), and in Nigeria, such methods are considered relatively effective (22). Nonetheless, physical control has to be intensive and regular and is thus ultimately expensive. The cost of clearing a hectare of weed is estimated to be between US \$2400–30,000 (23,24). Therefore, for a waterbody of more than a few tens of hectares, physical control is not a feasible long-term control option.

Chemical control is, worldwide, one of the most commonly used methods of macrophyte repression. Chemicals can be applied from the air, water, or land, and some degree of accuracy as to where the herbicide lands can be achieved. Chemicals such as diquat, glyphosate, amitrole, and the amine and acid formulations of 2,4-D are the most effective and commonly used chemicals against water hyacinth. Application needs to be undertaken by trained individuals using correct dosages and applied at appropriate stages of growth. When they are applied incorrectly, problems can be encountered, not least of which is killing nontargeted plant species such as aquatic macrophytes or agricultural crops (18). Herbicides themselves typically only cost in the region of US \$ 50 ha⁻¹ of treated area; however, application costs whether by plane, boat, or hand are in addition to this (25). Therefore, herbicide use is only viable over small areas; on the larger scale application, it can become prohibitively expensive. However, whatever the operational costs, if weeds grow faster than chemicals can be applied, then the control of water hyacinth is clearly not accomplished (11). Moreover, if the dead or dying weed is not removed, the rotting biomass can lead to localized deoxygenation, which can, in turn, lead to other detrimental impacts on the waterbody (19,26). Furthermore, although plants may appear to have been controlled, the problem may not have been solved if, as in the case of water hyacinth, seeds remain buried in the sediment and propagate some years later or, as is more typical, plants remaining in the marginal vegetation recolonize the waterbody. Repeated applications are therefore required (5).

The main problem associated with herbicide use, and indeed the reason they are not used as widely as they might be, is the potential effect, real or perceived, of chemicals on humans either directly through ingestion or indirectly through reduced income. Many affected rivers and lakes are major sources of drinking water, and therefore the problem of ingestion is clear. Also, the sites most affected by water hyacinth, and therefore most in need of herbicide application, are typically adjacent to lakeshore communities and farmlands, which increases the risk of destroying nontargeted plants and crops. Furthermore, fear concerning the impact on native biota, not just the loss of nontargeted plants, but also the potential for bioaccumulation of contaminant dioxins through the food chain, is also an area of concern for

governments considering their own populations health and the sale of fish to international exporters (27). Lake Victoria, for instance, supports an extremely valuable export fishery that earns critical foreign exchange for the riparian countries. The European Union, which is a major importer of fish from the lake, made it very clear that widespread application of chemicals could threaten the marketability of the fish. Finally, if biological control is used (see below), the application of herbicides may have impacts on the biological agents. Although experiments have found that diquat, glyphosate, and 2,4-D do not increase mortality rates of *Neochetina eichhorniae* (Fig. 3), they did move away from treated plants to untreated ones (28). In addition, researchers found that *N. bruchi* mortality was increased by several of the commonly used herbicides (29).

On the whole, chemical control, if undertaken correctly, can be very effective in the short term, but as with physical control, the use of herbicides is too expensive for repeated use on large expanses of water.

Biological control involves the use of host-specific agents that are naturally occurring enemies of invasive weeds in their native bioregion. The United States Department of Agriculture initiated biological control in Argentina in 1961, and since then, weevils, moths, and fungi have been tested for host specificity (30). Host-specific agents have now been successfully used to control water hyacinth in a variety of localities around the globe (5,11,31). Indeed, by 1998 a total of 7 agents had been released in 33 countries (32). Biological control of water hyacinth has therefore become very popular, and two weevils in particular, *N. eichhorniae* and *N. bruchi*, have been used extensively and successfully in North America, Asia, Australia, and Africa.

In the long term, once the weed is under control, a natural flux between agent and weed should exist, thus

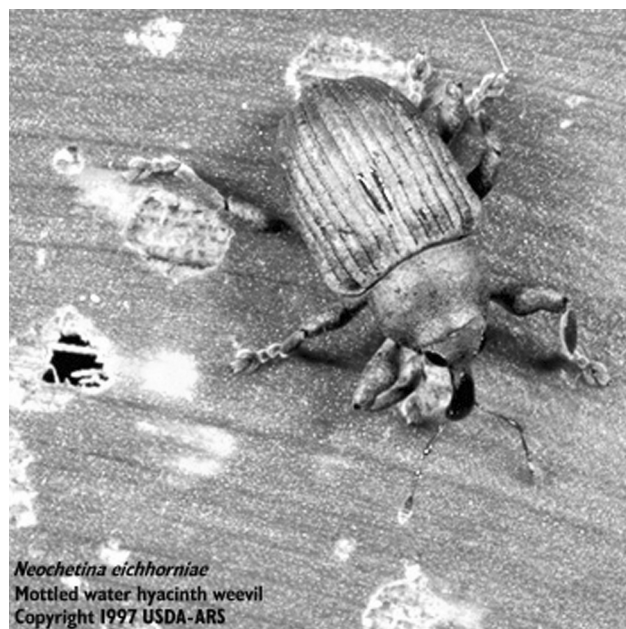


Figure 3. Biological control using the mottled water hyacinth beetle—*Neochetina eichhorniae*.

keeping in balance the density of weed and numbers of weevils present (33), although, as with all dynamic systems, fluctuations in both populations will occur (34). Biological control can provide an environmentally sensitive, cost-effective, and permanent solution to the problem of water hyacinth. However, the two drawbacks, assuming that host specificity is guaranteed, is that biological control usually takes between 3 to 5 years for the weevil populations to reach a level from which control can be achieved (5,25) and, moreover, it does not always work (35–37).

Overall, of most use in the control of water hyacinth is an integrated method of control. Integrated control employs all three of the above control methods in an overall management strategy. Each of the three is tailored to suit a specific problem, and the extent to which each is used varies with location. In general, however, mechanical and chemical controls are used initially to bring short-term benefits, whereas the biological approach, taking longer to implement and develop, follows after the first two control strategies (11,25). In this regard, maintaining weevil-rearing centers is an essential feature of active management to ensure that weevil populations remain at an adequate level for control.

One other type of control exists that is of key importance but yet is not mentioned as often as the above control methods—the control of nutrient inputs. The levels of available nitrogen and phosphorus have often been cited as the most important factors in limiting water hyacinth growth (38–43). Indeed, a Commonwealth Science Council survey of exotic floating weeds throughout Africa (34) recommended “that action be taken at a national and sub-regional level to reduce nutrient input (particularly from urban and industrial waste) into water bodies, and to develop land-use schemes for long-term utilisation and sustainable development of watersheds.” However, in 1993, at a symposium on water hyacinth control, only three papers from 13 country and regional reports mentioned the importance of managing nutrient inputs through improved watershed management. In 1998, during a global working group on biological and integrated control of water hyacinth, in a session dedicated to integrated management, only two out of the 13 papers presented identified the importance of nutrients in controlling water hyacinth. A similar scenario was seen in 2000, at the second meeting of the global working group.

It is understandable and necessary that from within affected countries scientists and practitioners of weed control turn to mechanical, chemical, and biological means of short-term control and management. Moreover, in the short term, all biological possibilities should be vigorously explored, as appropriate insects are available and have been used elsewhere often to good effect. Yet the only real long-term solution to the water hyacinth problem is not biological control but rather the control of anthropogenic eutrophication through sustainable management of watersheds and resources. The reliance of developing nations on biological control as a long-term solution rather than sustainable management

and development of their treasured resources is of major concern.

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WATER QUALITY IN PONDS

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INTRODUCTION

A pond is a body of water usually smaller than a lake, encircled by vegetation, and generally shallow enough for sunlight to reach the bottom. Rotted plants can grow in any spot within the pond. Widespread plant growth, such as pickerelweed, spatterdock, and cattails, is quite common in and around ponds and aids in creating a habitat for various forms of animal life, including insects (e.g., damselflies and

dragonflies), bullfrogs, greenfrogs, red winged blackbirds, marsh wrens, and bats.

Temporary ponds, that is, ponds that are dry for part of the year, are especially interesting and support a unique community. Organisms in such ponds must be able to survive in a dormant stage during dry periods or be able to move in and out of ponds, as can amphibians and adult aquatic insects. The temporary pond is a favorable place for those organisms adapted to it because interspecific competition and predation are reduced (1).

The chemical composition of any pond water ultimately depends on those elements and compounds in solution, in suspension, and those accumulating along the pond bottom, most of which come secondarily to the ponds from runoff. Other substances are produced by chemical reactions between the different elements and/or compounds by vegetation growing in or around the ponds.

POND WATER QUALITY

Salts

Water in closed pond basins is unlike the waters of other ponds in two respects: (1) it continually experiences a change in chemical composition because (2) it continually increases its concentration of various salts, mainly due to a high evaporation rate (2). However, the increase in salts in a closed basin is not infinite. Certainly, as water flows into a dry pond basin, perhaps from the first rains of late summer or early fall, it carries the salts from rocks surrounding the lake basin and dissolves the surface salts of the playa, providing they have not been removed by deflation. As precipitation declines, generally after the early spring rains, evaporation becomes greater than inflow which then starts the inevitable process of concentration of the salts in solution. Salts concentrated in the waters of closed basins depend mainly on the rock type(s) of the surrounding basins and/or the chemical characters of any influents and, to a minor degree, on several inorganic ions in rainwater.

Pollutants

This catchall category of pollutants means anything added to the pond that is not wanted in the pond. Pollutants can consist of items that may or may not be harmful to human beings as well as fish, and they may or may not be visible. They may float on the surface, sink to the bottom, or dissolve in the water. They may come from outside the pond or from within the pond itself, for example, oil leaking from a submerged pump. Some pollutants are easy to identify and control and/or remove, such as leaves, pollen, and dead rats; some just add to the filter load if not removed but cause few other problems if not in excessive amounts, for example, bird droppings. Most harmful pollutants that dissolve in water are hard to identify or quantify. Surface water runoff that can enter the pond is often a major source of pollutants. That is why all ponds should be designed with a raised edge or at least some type of channel around them so that the surface water does not enter.

Dissolved Oxygen

Whenever air is in contact with water, whether through natural or artificial means, a transfer of oxygen from the air to the water takes place until the water becomes saturated. Plants under light convert carbon dioxide to oxygen in water. Fish, plants at night, and aerobic bacterial action consume the oxygen.

As the air components dissolve in the water, a point is reached where no more can be added. This point is called saturation. The saturation points are different for each of the gases and depend on several factors, but temperature is the most important. As the temperature increases, the water simply cannot hold as much of each type of gas. For oxygen, the approximate saturation level is 11.5 mg/L at 50°F, 9 mg/L at 70°F, and 7.5 mg/L at 90°F. Impurities added to the water (i.e., salt) decrease these saturation levels.

Water whose oxygen concentration is less than 3 mg/L will not generally support fish. When the concentration falls to about 3–4 mg/L, fish start gasping for air at the surface. Levels from 3–5 mg/L can normally be tolerated for short periods. Above 5 mg/L, almost all aquatic organisms can survive indefinitely, provided other environmental parameters are within allowable limits.

Surface oxygen from 11–15 mg/L creates a highly eutrophic condition. Depletion of dissolved oxygen in pond water can encourage the microbial reduction of nitrate to nitrite and sulfate to sulfide, giving rise to odor problems. It can also cause an increase in the concentration of iron (II) in solution (3).

Color

The color of pond water depends on the depth, the color of the surrounding area and sky, the scattered light rays coming from the water, and in most cases, absorption of scattered light rays by dissolved or suspended materials.

Pond water is typically blue for pure, clear water, and various shades of blue–green to a definite green, then into the yellows and browns; the various colors are produced by the innumerable contaminants (Table 1). In general, black pond water results from high organic content, and blue pond water results from an absence of contaminating materials. The color of most pond water exists because of light scattering by vibrating water molecules blue predominates because molecular scattering is greater for shorter wavelengths.

Ammonia

Ammonia (NH_3), in parts per million (ppm), is the first measurement to determine the “health” of a biological converter. Ammonia should not be detectable in a pond that has a “healthy” bioconverter. The ideal and normal measurement of ammonia is zero. When ammonia is dissolved in water, it is partially ionized depending upon the pH and temperature. Ionized ammonia is called ammonium and is not toxic to fish. As the pH drops and the temperature decreases, ionization and ammonium increase, which decreases the toxicity. The presence of ammonia in ponds at higher than geogenic levels is an important indicator of fecal pollution (5).

Table 1. Variation in Water Color^a

Color	Contaminants
Clear blue (pure water)	No contaminants of suspended materials; color due to molecular scattering
Bluish green	Suspension of blue-green algae, phytoplankton
Green	Colloidal CaCO ₃ (hard water)
Yellowish green	Suspended sulfur
Greenish yellow	High plankton content, excluding humic debris
Yellow	Diatoms
Yellow brown	Diatoms-fluorescent compounds, humic acids, carboxylic acids, organic matter
Gray	<i>Spirostomum</i>
Black	Organic debris, oxidation of ferrous iron, <i>Stentor</i>
Red to orange to purple	Suspended clays, ferrous hydroxide, <i>Oscillatoria rubescens</i> , <i>Euglena sanguinea</i> , <i>Haematococcus pluviolis</i> , <i>Dunaliella salina</i> , <i>Bacterium halobium</i> , purple sulfur bacteria, zooplankton.

^aReference 4.

Ammonia tends to block oxygen transfer from fish gills to the blood and can cause both immediate and long-term gill damage. The mucous-producing membranes can be destroyed, reducing both the external slime coat and damaging the internal intestinal surfaces. Fish suffering from ammonia poisoning usually appear sluggish, often at the surface as if gasping for air.

Ammonia is a gas primarily released from fish gills as a metabolic waste from protein breakdown and from some lesser secondary sources such as bacterial action on solid wastes and urea.

When ammonia is detected (assuming a pH of about 7.5),

1. Increase aeration to maximum. Add supplemental air if possible.
2. Stop feeding the fish if detected in an established pond.
3. For an ammonia level of 0.1 ppm, conduct a 10% water change out. For a level of 1.0 ppm, conduct a 25% change out.
4. Consider transferring fish if the ammonia level reaches 2.5 ppm.
5. Retest in 12 to 24 hours.

Nitrite

Nitrite is produced by the autotrophic *Nitrosomonas* bacteria that combine oxygen and ammonia in the bioconverter and to a lesser degree on the walls of the pond.

Nitrite has been termed the invisible killer. The pond water may look great, but nitrite cannot be seen. It can be deadly, particularly to the smaller fish, in concentrations as low as 0.25 ppm. Nitrite damages the nervous system, liver, spleen, and kidneys of fish.

Whenever 0.25 ppm of nitrite or more is detected in a pond,

1. Increase aeration to maximum. For a nitrite level of 1 ppm or greater, add supplemental air, if possible.
2. Discontinue use of any UV sterilizers and ozone generators.
3. For a nitrite level less than 1 ppm, conduct a 10% water change out, and add 1 pound of salt per hundred gallons of changed water.
4. For nitrite levels of 4.0 or greater, consider transferring fish.

Nitrate

Nitrate is produced by the autotrophic *Nitrobacter* that combine oxygen and nitrite in the bioconverter and to a lesser degree on the walls of the pond. The nitrate concentration in surface water is normally low but can reach high levels as a result of agricultural runoff, refuse dump runoff, or animal wastes. Concentrations from zero to 200 ppm are acceptable. Ammonia and nitrite are toxic to fish, but nitrate is essentially harmless.

The nitrate concentration is controlled naturally through routine water change outs and to a lesser degree through plant/algae consumption.

Temperature

The temperature of the water in a pond is seldom constant; variations depend on depth, season, and geography. Pond water is heated by absorption of solar radiation. Events generally happen faster at higher temperature and in smaller ponds. Over normal temperature ranges, biological activity doubles for each 10°C rise in temperature. The toxicity of ammonia increases as the temperature rises, and the amount of dissolved oxygen that the water can hold decreases.

Direct sunlight during the day can cause the temperature to rise higher, and heat loss on clear nights can cause the temperature to drop lower than that of shaded ponds. A clear night sky can absorb a large amount of heat from a small pond and actually drive the pond temperature below air temperature.

Plant Nutrients and Cultural Eutrophication

Pond water clarity (transparency) is affected by sediments, chemicals, and the abundance of plankton organisms; it is a useful measure of water quality and water pollution. Ponds that have clear water and low biological productivity are said to be oligotrophic. By contrast, eutrophic waters are rich in organisms and organic materials. Eutrophication, an increase in nutrient levels and biological productivity, often accompanies successional changes in ponds. Surrounding runoffs bring in sediments and nutrients that stimulate plant growth. Over time, ponds often fill in, becoming marshes or even terrestrial biomes. The rate of eutrophication depends on water chemistry and depth, volume of inflow, mineral content of the surrounding watershed, and the biota of the pond itself (6).

Human activities can greatly accelerate eutrophication, an effect called cultural eutrophication. Cultural eutrophication is caused mainly by increased nutrient input into a waterbody. Eutrophication produces “blooms” of algae or thick growth of aquatic plants stimulated by elevated phosphorus or nitrogen levels (Fig. 1). Bacterial populations then increase, fed by larger amounts of organic matter. The water often becomes cloudy or turbid and has unpleasant tastes and odors. Cultural eutrophication can accelerate the “aging” of a waterbody enormously over natural rates. Ponds that normally might exist for hundreds of years can be filled in a matter of decades.

pH

The pH of water is a measure of the acid–base equilibrium and, in most natural waters, is controlled by the carbon dioxide–bicarbonate–carbonate equilibrium. An increased carbon dioxide concentration in pond water will therefore lower pH, whereas a decrease causes it to rise. Temperature also affects the equilibria and the pH. In pure water, a decrease in pH of about 0.45 occurs as the temperature is raised by 25 °C.

A pH measurement helps to determine if water is suitable for fish. The pH should normally be between 7.0 and 8.5; it is probably acceptable if it is anywhere between 6.0 and 9.0. Although most fish could tolerate a pH as low as 5.0, bioconverter bacteria are subject to damage. Long-term conditions above pH 9.0 can cause kidney damage to some fish.

A sudden change of a half or more pH unit in an established pond is an indication that something happened, and the cause should be determined. An increasing pH trend in a pond is normally caused by lime leaching out of concrete and to a lesser degree by concentration due to evaporation and decomposing organic matter. Decreasing pH is due primarily to bacterial action that releases acidic compounds. Established ponds normally maintain their equilibrium pH if sludge and decaying organic material are routinely removed.



Figure 1. Eutrophic pond. Nutrients from agriculture and domestic sources have stimulated the growth of aquatic plants. This reduces water quality, alters species composition, and lowers the pond's aesthetic values.

Scheduled water change outs (10% per week for small pond, less for larger ponds) are also helpful. At pH extremes approaching 4 or 11, remove any remaining fish.

Saprotrophic Organisms

Aquatic bacteria, flagellates, and fungi are distributed throughout a pond, but they are especially abundant in the mud–water interface along the bottom where bodies of plants and animals accumulate. A few of the bacteria and fungi are pathogenetic, attack living organisms, and cause disease, but the great majority begins attack only after an organism dies. When temperature conditions are favorable, decomposition occurs rapidly in a body of water; dead organisms do not retain their identification for very long but are soon broken up into pieces, consumed by the combined action of detritus-feeding animals and microorganisms, and their nutrients are released for reuse (1).

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WATER TURBINE

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Water turbine, also called a hydraulic turbine, is a rotary machine that converts the power of flowing water into usable electronic or mechanical power. The word *turbine* is derived from the Latin word for “whirling” or a “vortex”. Water turbines were developed in the nineteenth century during the Industrial Revolution, using scientific principals and methods, for industrial power. Now they are mostly used for hydropower generation, which typically requires a construction of a dam to form a reservoir. They are one of the cleanest producers of power, replacing the burning of fossil fuels and eliminating nuclear waste. They use a renewable energy source and are designed to operate for decades. They produce significant amounts of the world's electrical supply. On the other hand, there are some negative consequences; by their nature, water

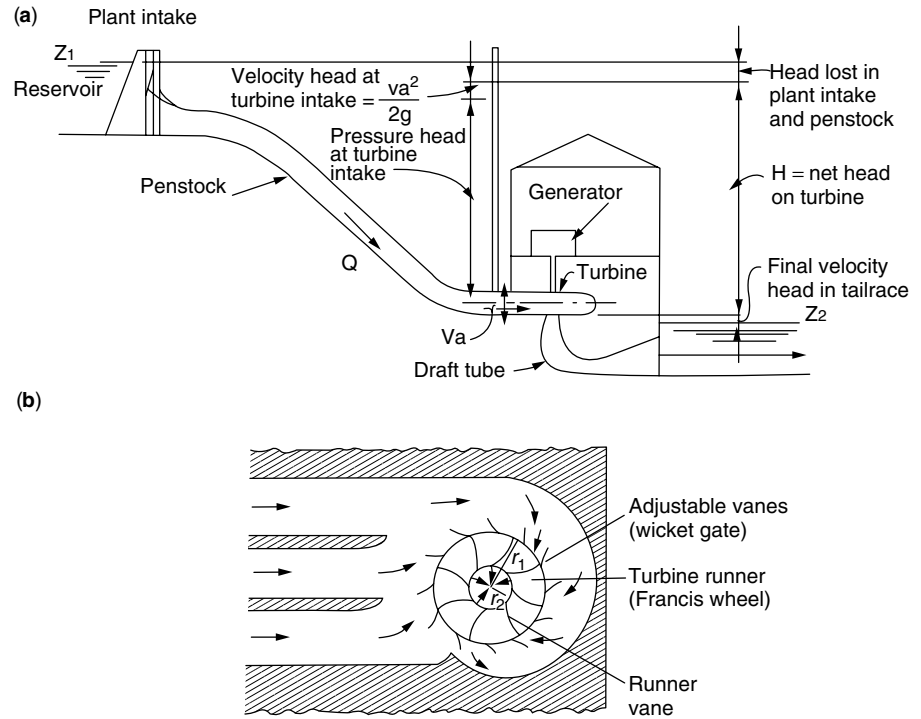


Figure 1. Hydropower arrangement for reaction turbine. (a) Hydropower arrangement from upstream reservoir to downstream river. (b) Plane view of reaction turbine installation.

turbines interrupt the natural ecology of rivers, stopping migrations of fishing.

A water turbine is usually connected to an elevated reservoir (Fig. 1) through large pipe (penstock), which transfers potential energy of water into kinetic energy. Total hydraulic head (H) available for a turbine is equal to the elevation difference of water surfaces between upstream reservoir (Z_1) and downstream river (Z_2) minus any friction and local energy losses (HL) through penstock, inlet, and turbine system. The electric power generated by a water turbine is

$$P = \frac{\gamma Q H \eta}{1000} = \frac{\gamma Q (Z_1 - Z_2 - HL) \eta}{1000} \text{ (SI units)}$$

$$P = \frac{\gamma Q H \eta}{550} \text{ (English units)} \quad (1)$$

where η is the turbine efficiency and P is the electric power in kilowatts for SI units and in horsepower for English units. Large industrial water turbines typically operate at efficiencies greater than 90%, which is much higher than overall efficiency of thermal power generation (on the order of 30%). From Equation 1, it indicates that decrease of energy losses in penstock and inlets results in an increase of power generation by a turbine. Power generation depends on elevation differences of water surface in upstream reservoir and downstream river, which fluctuate over season and over year. Water turbines can run continuously to provide electricity generation but typically vary power output over a season. Water turbine system can be relatively easy to turn on and turn off in comparison with a thermal or nuclear power system; therefore, some of water turbines only provide electricity during high-power demand periods (e.g., during the day). When the power network has extra power during night,

e.g., developed by thermal and nuclear power plants, extra power can pump water from a downstream reservoir to an upstream reservoir and can be used for future power generation. A pump turbine is used for the above purpose and is a hydraulic machine that operates as a pump in one mode of operation and as a turbine in a different mode. A turbine and generator can supply power under varying load demands at a constant rotational speed by adjusting control wicket gates (Fig. 1), but overall efficiency and performance of turbine will vary somewhat with the gate opening.

Water turbines are divided into two groups: impulse turbines and reaction turbines, according to their hydraulics action. Impulse turbines are the modern forms of water wheels that have been used for thousands of years for industrial power. The impulse turbine is typically used for high-head hydropower installation (e.g., $H > 1500$ ft) but for relative small discharge. In the impulse turbine, jets of water issuing from one or several nozzles impinge on vanes (buckets) of the turbine wheel (also called runner), which reverses the flow of water, and the resulting change in momentum (impulse) causes the turbine to spin. A nozzle in reducing flow area is to create a high speed of water jet (V_j). The fluid momentum equation can determine what forces can be developed on buckets because of a water jet. When the speed of turbine runner is half of the speed of water jet, the optimum power is developed by an impulse turbine:

$$P = \gamma Q \left(\frac{V_j^2}{2g} \right) \quad (2)$$

Reaction turbines are typically used for low-head (1–100 ft) and medium-head (90–1500 ft) hydropower

plants with moderate-to-high discharges. For reaction turbines, only part of the available head (H) is converted into kinetic energy, and a substantial part remains as a pressure head that varies throughout the turbine water passage. Reaction turbines include radial-flow Francis turbines for medium-head and axial-flow Kaplan turbines with or without adjustable runner blades for low-head hydropower plants. In 1849, James B. Francis conducted sophisticated tests, developed engineering methods for water turbine design, and improved the efficiency of reaction turbine to over 90%. The Francis turbine, named for him, was the first modern water turbine. It is still the most widely used water turbine in the world today. The Kaplan turbine, a propeller type machine, was invented around 1913 by Victor Kaplan. It was an evolution of the Francis turbine, but it revolutionized the ability to develop low-head hydropower plants. S-type and bulb turbines are other axial flow reaction turbines used for tidal rivers or estuaries that have only several feet of water surface elevation differences. They can generate electric power for tidal flow in two different flow directions.

As the swirling mass of water spins into a tighter rotation of a reaction turbine, it tries to speed up to conserve energy. This property acts on the runner; in addition to the water's falling weight, water pressure decreases to zero as it passes through turbine blades and gives up its energy to make the turbine runner to rotate continuously. Francis turbines have fixed runner blades and use a series of adjustable wicket gates (Fig. 1) to control flow into turbine runner to achieve high efficiency under various flow rate and available head conditions. Flow for the Francis turbine enters the runner in a radial direction, whereas flow of the Kaplan turbine enters the runner in a direction parallel to the turbine's rotational axis. A Kaplan turbine can have adjustable blades for its runner to improve turbine efficiency.

The specific speed (n_s or N_s) of a turbine characterizes the turbine's shape and performance in a way that is not related to its size, which allows a new turbine design to be scaled from an existing design of known performance. The specific speed is the main criteria to select the correct turbine type for matching a specific hydropower development.

$$n_s = \frac{nQ^{1/2}}{g^{3/4}H^{3/4}} \quad \text{or} \quad N_s = \frac{NP^{1/2}}{H^{5/4}} \quad (3)$$

The specific speed n_s is a dimensionless parameter; n is the rotational speed of turbine runner in rps (revolution per second), Q is the discharge passing through turbine in ft^3/s or m^3/s , H is the available head in ft or m , and g is the acceleration of gravity as 32.2 ft/s^2 or 9.81 m/s^2 . The specific speed N_s requires N in rpm (revolution per minute), P in horsepower, and H in feet. The specific speed of a turbine is given by the manufacturer (along with other ratings) and will always refer to the point of maximum efficiency. These allow accurate calculations to be made of the turbine's performance for a range of heads and flows. Given a flow and a head for a specific hydropower development site, and the rpm requirement of the generator, one can calculate the specific speed. The

result is the main criterion for turbine selection; e.g., impulse turbines have N_s less than 20; Francis turbines have N_s about from 15 to 85; and Kaplan turbines have N_s about from 65 to 195 (1).

What should be the diameter of a turbine runner (D)? It is dependent on the turbine's speed ratio ϕ :

$$D = \frac{60 \times \phi \sqrt{2gH}}{\pi N} \quad \text{or} \quad \phi = \frac{u}{\sqrt{2gH}} = \frac{\pi DN/60}{\sqrt{2gH}}$$

where u is the linear rotational speed of a turbine runner and N is the rotational speed in rpm. For a 60-cycle frequency, N is equal to 7200 divided by the number of poles (n) in the generator, which must be an even number. In Europe and South America, where the frequency is 50 cycles/second, the speed N is given as $6000/n$. The ranges of speed ratios for maximum efficiency for the different types of turbines are 0.43–0.47 for impulse turbine, 0.5–1.0 for Francis turbine, and 1.5–3.0 for Kaplan propeller turbine.

Turbines and pumps are susceptible to cavitation. Cavitation can damage the turbine runner and affect turbine performance. The region of a turbine where cavitations are most likely to occur is on the downstream side of the turbine impeller blades. The cavitation index (σ) can evaluate the possibility of turbine cavitation:

$$\sigma = \frac{(P_{atm} - P_v)/\gamma - (Z_t - Z_2)}{H}$$

where P_{atm} and P_v are the local absolute atmospheric pressure and vapor pressure of water and Z_t and Z_2 (Fig. 1) are elevations of the downstream side of the turbine runner and tailwater. Lower values of σ indicate a greater tendency for cavitation. The critical sigma (σ_c) for different types of turbines are obtained experimentally and typically provided by turbine manufacturers. When σ is greater than σ_c , there is no cavitation; otherwise cavitation is highly possible.

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WETLANDS: USES, FUNCTIONS, AND VALUES

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Throughout history, wetlands have played an important role in human development. They have brought benefits, but also caused difficulties for people. Their perceived value to society has varied from place to place and has changed over time.

In many arid and semiarid regions of the world, societies have flourished because of the water and other resources provided by wetlands. Many of the great civilizations of history have benefited from wetlands. For example,

the Maya, Inca, and Aztec civilizations in Latin America; the Khmer civilization of Angkor in the lower basin of the Mekong; and the civilizations of the Nile, Niger, and Mesopotamia were all dependent on wetland resources. In contrast, in wetter parts of the world, wetlands were for many centuries widely thought of as unproductive wastelands often associated with disease, difficulty of access, and danger. In East Anglia in the United Kingdom, poor drainage within an area of approximately 3,870 km² of fens meant serious delays in sowing crops and loss of harvests because of flash floods. Furthermore, illnesses and epidemics were believed to originate in the bad air ("miasmas") of swamps and bogs. A common perception was that wetlands were "awful reservoirs of stagnated water, which poisons the circumambient air for many miles around about and sickens and frequently destroys many of the inhabitants" (1).

In Europe, land drainage schemes were initiated as early as the thirteenth century to dry out wetlands and improve their agricultural productivity. Major drainage schemes in Britain in the seventeenth century converted thousands of square kilometers of fens and marsh into, what is today, some of the most productive farmland in the country (2). Similar schemes were implemented across Europe and elsewhere in the world. During the twentieth century, technological advances coupled with greater intensity of agriculture, as well as urban and industrial development, increased the pace of change. Alteration in the water balance and the quality of water entering wetlands can result in desiccation and degradation of wetland habitat. Although very difficult to quantify, it is estimated that globally more than half of all wetlands have been destroyed as a consequence of human activities (3).

In recent years, it has been realized that wetlands are valuable ecosystems. It is now understood that they maintain environmental quality, sustain livelihoods, support immense biodiversity, and have important recreational and aesthetic qualities.

Wetlands continue to play an important role in sustaining the livelihoods of many millions of people, particularly in Africa and Asia. For example, the valley bottom wetlands in Africa are of major importance in agricultural systems. Saturated during the wet season, they remain wet into the dry season effectively extending the growing season. As a result, these wetlands act as a key resource for cultivators and pastoralists by providing a source of arable land or grazing throughout the dry season and during droughts. Similarly, in Bangladesh, many thousands of cattle that graze on floodplains during the dry season are watered from perennially flooded areas known locally as *beels*.

Insight into their value has led to a radical change in attitude toward wetlands and has been the primary reason for a range of national and international conservation policies. Throughout Europe, the United States, and many other countries, considerable efforts are now being made to protect and even recreate wetlands (4).

WETLAND FUNCTIONS

Wetlands are among the most productive ecosystems in the world, characterized by high primary productivity

and a multiplicity of habitats that enables them to support a large diversity of life. The wide range of natural phenomena that occur within wetlands makes them valuable in socioeconomic as well as ecological terms.

Wetland values arise from the ecological functions they perform. Ecological functions are the result of the physical, chemical, and biological processes that occur through the interaction of the biotic (i.e., flora and fauna) and abiotic (i.e., soil and water) components that comprise an ecosystem. The exact nature of the functions is unique to any particular wetland and depends on site-specific factors, including climate, geology, topography, and the wetland's location within a catchment. However, it is now recognized that biodiversity per se, plays an important role in maintaining wetland processes (e.g., condensation and evapotranspiration) across a range of temporal and spatial scales (5).

Not all wetland functions are useful to people. Some are harmful. For example, provision of habitat for mosquitoes and other insects that transmit illnesses is a function of many wetlands that throughout history has resulted in incalculable human suffering. However, many wetland functions do enhance human well-being, and it is now widely accepted that, overall, the advantages of wetlands outweigh their disadvantages.

The *physical* benefits human populations derive from wetlands comprise "goods" and "services." Goods are the natural resources that occur within a wetland and can be exploited by people. Examples include plants and wildlife used for food, trees and reeds used for construction materials, and vegetation used for medicine. Services are the nonresource benefits that people gain from the interaction of wetland processes. Examples include hydrologic, water quality, and geomorphological functions, such as flood reduction, nutrient transformation, and sediment retention. A summary of some of the physical benefits that human societies gain from wetland functions is presented in Table 1.

The high productivity of wetland ecosystems is reflected in the high human populations that they can support. In Africa and Asia, hundreds of millions of people depend on wetlands for water, rice, fish, shrimp, and other food production. In many tropical wetlands, fisheries production is extremely high. The lower Mekong Basin supports an inland fishery whose annual yield is approximately 500,000 tons, and in Indonesia, half the commercial fish catch comprises species dependent on estuarine wetland systems (6). The freshwater and estuarine wetlands of the Nile, Congo, Riffiji, Tana, Niger, and Zambezi river catchments in Africa each sustain the livelihoods of many hundreds of thousands of people by providing fisheries; dry season grazing for cattle; water for domestic use; and agriculture and plants for traditional medicines, mat production, and firewood. Similarly the great riverine and floodplain systems of South and Central America support large numbers of subsistence and commercial fisheries. It is estimated that 75% of the commercial fish catch of the Amazon River depends on food chains originating in seasonally flooded forest wetlands (6).

Table 1. Examples of Wetland Functions that Benefit Human Populations

Wetland Function	Use by Society	Examples
Storage of precipitation and runoff	- Water supply - Flood protection - Agriculture	- In Florida, U.S., approximately 5 million people, including many in the city of Miami are supplied with water from the wetlands of the Florida Everglades. 10,000 people living in the vicinity of Laguna El Jocotal, a shallow floodplain lake in El Salvador, depend on the lake for their dry season water supply. - A flood prevention value of about US\$13,500 per hectare per year has been attributed to wetlands in the Charles River catchment in Massachusetts, U.S. - In West Africa, the shallow groundwater and elevated soil moisture of about 5 million hectares of inland valley wetlands are used extensively in crop cultivation.
Groundwater discharge	Water supply	8% of the discharge of the Mzima springs in Tsavo National Park, Kenya, is diverted to supply the coastal city of Mombasa.
Groundwater recharge	Water supply	The Hadejia–Nguru wetlands in Nigeria contribute to the recharge of an aquifer used for water supply by approximately 1 million people who live in the region.
Sediment retention	Shoreline protection	- Gravels underlying riverine forest wetlands adjacent to the Danube supply a large proportion of the water to the 1.6 million people of Vienna, Austria. - The mangroves of the Indus delta protect the coastline from wind and currents and shelter Port Qasim, Pakistan's second largest port.
Nutrient transformation	- Wastewater treatment - Protection of water resources	12,000 hectares of wetland to the east of Calcutta, India, are used to treat approximately 68 million liters of sewage each day. In 10 days, it is transformed into "sweet water" suitable for irrigation. - A riparian wetland system on the River Lambourn, UK, removed up to 85% of the nutrients in agricultural runoff before it reached the river.
Biomass production (e.g., wood, grass, fish)	Food	- Local people use the fruits, seeds, tubers, roots, and leaves of around 200 plants from the wetlands surrounding Lake Chilwa in Malawi. - The wetlands of the Kafue floodplain in Zambia support fisheries averaging 6,500 tons per year.
Maintenance of biodiversity	- Building and construction materials - Medicine	- In Matang Forest Reserve, Malaysia, 40,000 hectares of mangroves annually yield timber worth US\$9 million. - The St. Regis Mohawk tribe in the United States uses some 85 wetland plants for medicine, including <i>Acorus calamus</i> used for chills, fevers, and coughs and <i>Lobelia cardinalis</i> used as a blood purifier. - In Peninsular Malaysia, the sale of medicinal plants originating in wetlands can earn households up to US\$ 80 per month.
Chemical cycling	Gas regulation	- Wetlands are important in the biogeochemical cycling of carbon dioxide, methane, and hydrogen sulfide. - Globally, wetland peat deposits occupy just 3% of the world's land area but store 16–24% of the planet's soil carbon pool.

Some wetlands play an important role in maintaining or improving water quality. The large number of chemical and biological processes that take place within wetlands (including nitrification, denitrification, ammonification, and volatilization) may convert heavy metals to insoluble forms and remove nutrients from the water. In Uganda, the Nakivubo papyrus swamp receives semitreated effluent from the sewage works of the capital city, Kampala. During the passage of the effluent through the wetland, the papyrus vegetation absorbs nutrients and the concentration of pollutants is reduced before the water enters Lake Victoria, the principal water source for the city

(7). In Malaysia, the North Selangor peat swamp protects the quality of water used for water supply and irrigation by promoting sedimentation, nutrient uptake, toxic metal removal, and pH buffering (8).

People also gain *nonphysical* benefits from wetland functions. These are associated with spiritual enrichment, cognitive development, and aesthetic experience. For example, the culture of the Lozi people in western Zambia has developed as a consequence of the hydrologic dynamics of the Barotse floodplain. Each year, cultural ceremonies are conducted when the people move out of the floodplain during periods of high water and back again

when water levels fall. Such ceremonies are important in enhancing the social cohesiveness of communities. Elsewhere, wetlands such as the Florida Everglades in the United States, the Okavango Delta in Botswana and the Caroni Swamp in Trinidad are major tourist attractions, not only enriching the lives of visitors, but also generating income for others.

Wetlands support a rich diversity of flora and fauna. Some animals live permanently in wetlands (e.g., the manatee, swamp tortoise, and lechwe antelope); others use them as refuges and migrate to them when conditions of drought or food scarcity exist elsewhere or at particular stages of their life cycle. Wetlands are especially important for migratory birds. Of the 300 species of birds that migrate between Europe and Africa each year, a large number depend on wetland habitats to support them enroute. The Patanal wetland in South America provides habitat for some 865 species of birds, and the Okavango Delta and Kafue Flats in Africa each support 400 species. It is estimated that the Inner Delta of the Niger Delta hosts up to 1.5 million birds each year (9). The beauty and naturalness of what are perceived to be pristine and increasingly rare ecosystems, and the variety of wildlife that they support, are important and of value to increasing numbers of people worldwide.

Hence, wetlands bring a wide variety of both tangible and intangible benefits to a huge number of people. The way in which they do so is complex, multifunctional, and very often not obvious. Many groups of people benefit from wetlands, and very often the benefits are experienced far beyond the boundaries of the wetland itself. Many wetland contributions to human welfare accrue without people being aware of them. As a consequence, wetland functions have in the past often gone unrecognized in development and resource planning and management.

WETLAND VALUES

The values attributed to wetlands depend largely on social perceptions of the uses and the benefits to be gained from them. The view that wetlands are wastelands arose in countries where agricultural productivity is not usually limited by water scarcity, at a time when rapidly rising populations required increased food production and there was very little understanding of the benefits provided by wetlands.

Today, in the developed countries of the world, the situation is very different. The ecosystems that people most depend on are those in which the functions, composition, and structure are far removed from natural systems. High levels of social and economic security have been obtained through application of complex technologies and considerable alteration of natural ecosystems, including wetlands.

These societies are now sufficiently prosperous that they can contemplate a large number of options in the way that they manage ecosystems, and paradoxically, many of them are now placing very high value on the aesthetic quality and biodiversity functions of their remaining wetlands.

In contrast, many millions of people in much of the developing world remain directly dependent on ecosystems that are in a much more natural state. For many individuals, wetland functions provide the most important contributions to their livelihoods and welfare by providing food supplies, medicines, income, employment, and cultural integrity. Such communities often have limited alternative livelihood options, and this makes them particularly vulnerable to changes in the condition of the natural environment on which they depend.

As a consequence of these different perspectives, wetland values were, until recently, very different. In the developed world, the primary focus was on conservation of wetland "wilderness" for recreation and to maintain wildlife habitat, especially for waterfowl. In much of the developing world, the emphasis was on the utility of wetlands, in particular their role in food production and livelihood support. However, there is now a widespread convergence of views brought about by the realization of two factors. First, that although the natural resources they provide may not be as important in the developed world, wetlands, nevertheless, support or protect a wide range of economic activities: for example, through their role in flood reduction and water purification; second, that social and economic factors are the main reasons for wetland loss; poverty and environmental degradation are inextricably intertwined. Consequently, sustainable development is a prerequisite for successful conservation.

This convergence of views is reflected in the widely promulgated "wise use" policy of the Convention on Wetlands of International Importance especially as Waterfowl Habitat—commonly referred to as the Ramsar Convention. This states that "the wise use of wetlands is their sustainable utilisation for the benefit of humankind in a way compatible with the maintenance of the natural properties of the ecosystem" (10). Explicit in the guidelines of the Ramsar Convention is the need to identify the values that local people, who use wetlands directly, can bring to all aspects of wetland wise use.

The site-specific nature and diversity of wetland functions makes quantification of wetland values extremely difficult. However, the wise use of wetlands requires a way of comparing the various benefits and consequences of development options on wetland functions. Traditionally, valuation techniques were strictly financial. However, it is impossible to assess all the issues relevant to wetlands using money as the sole determinant in the assessment process. Consequently, in recent years, economists have developed a number of environmental valuation techniques that enable assessment of nonmonetary impacts (11). Such methods, for example, multicriteria analysis, enable assessment of nonmonetary values and are being increasingly used to evaluate alternatives in wetland management.

Management of environmental resources, including wetlands, will be a key human endeavor in the twenty-first century. The biotic impoverishment and disruption of wetland functions caused by past decisions and continuing mismanagement severely constrain the options for future management. If the benefits wetlands bring are to be used more wisely in the future, there is need for much

greater insight into the dynamics of wetland functions and the links between wetlands and human society. Only through such understanding will the true value of wetlands be determined.

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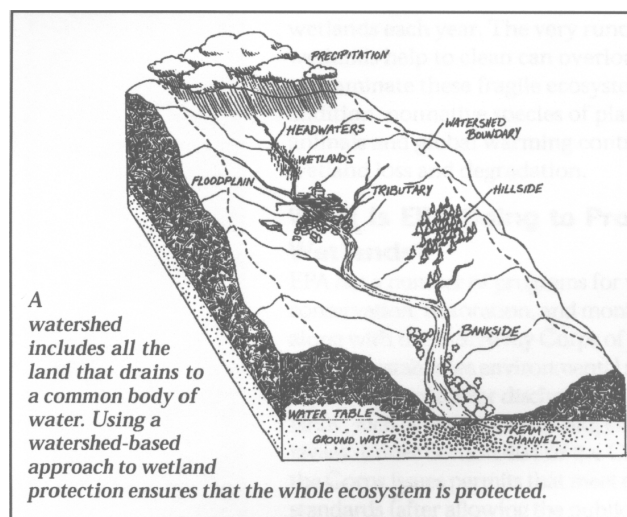
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WETLANDS OVERVIEW

U.S. Environmental Protection Agency—Office of Water, Office of Wetlands, Oceans and Watersheds

WHAT IS A WETLAND?

Although wetlands are often wet, a wetland might not be wet year-round. In fact, some of the most important wetlands are only seasonally wet. Wetlands are the link between the land and the water. They are transition zones



where the flow of water, the cycling of nutrients, and the energy of the sun meet to produce a unique ecosystem characterized by hydrology, soils, and vegetation—making these areas very important features of a watershed. Using a watershed-based approach to wetland protection ensures that the whole system, including land, air, and water resources, is protected.

Wetlands found in the United States fall into four general categories—marshes, swamps, bogs, and fens. Marshes are wetlands dominated by soft-stemmed vegetation, while swamps have mostly woody plants. Bogs are freshwater wetlands, often formed in old glacial lakes, characterized by spongy peat deposits, evergreen trees and shrubs, and a floor covered by a thick carpet of sphagnum moss. Fens are freshwater peat-forming wetlands covered mostly by grasses, sedges, reeds, and wildflowers.

GOOD NEWS

Often called “nurseries of life,” wetlands provide habitat for thousands of species of both aquatic and terrestrial plants and animals. These nurseries support the critical



Two-thirds of the 10 million to 12 million waterfowl of the continental United States reproduce in the prairie pothole wetlands of the Midwest, and in the winter millions of ducks like these can be found in the wetlands of the south-central United States

developmental stages for many plants and animals. Although wetlands are best known for being home to water lilies, turtles, frogs, snakes, alligators, and crocodiles, they also provide important habitat for waterfowl, fish, and mammals. Migrating birds use wetlands to rest and feed during their cross-continental journeys and as nesting sites when they are at home. As a result, wetland loss has a serious impact on these species. Habitat degradation since the 1970s has been a leading cause of species extinction.

Living systems cleanse water and make it fit, among other things, for human consumption.

Elliot A. Norse,
in R.J. Hoage, ed.,
Animal Extinctions, 1985

The nation behaves well if it treats the natural resources as assets which it must turn over to the next generation increased, and not impaired, in value.

Theodore Roosevelt, 1907

Wetlands do more than provide habitat for plants and animals in the watershed. When rivers overflow, wetlands help to absorb and slow floodwaters. This ability to control floods can significantly prevent property damage and loss and can even save lives. Wetlands also absorb excess nutrients, sediment, and other pollutants before they reach rivers, lakes, and other waterbodies. They are great spots for fishing, canoeing, hiking, and bird-watching, and they make wonderful outdoor classrooms for people of all ages.

BAD NEWS

Despite all the benefits provided by wetlands, the United States loses about 60,000 acres of wetlands each year. The very runoff that wetlands help to clean can overload and contaminate these fragile ecosystems. In addition, nonnative species of plants and animals and global warming contribute to wetland loss and degradation.



This forested wetland on the Chincoteague National Wildlife Refuge on Virginia's Eastern Shore is part of the Atlantic flyway, where shorebirds and waterfowl rest before they migrate south for the winter



A freshwater pool at Assateague National Seashore in Virginia

WHAT IS EPA DOING TO PROTECT WETLANDS?

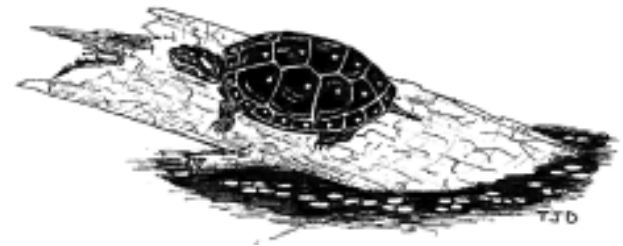
EPA has a number of programs for wetland conservation, restoration, and monitoring. EPA, along with the U.S. Army Corps of Engineers (Corps), establishes environmental standards for reviewing permits for discharges that affect wetlands, such as residential development, roads, and levees. Under Section 404 of the Clean Water Act, the Corps issues permits that meet environmental standards (after allowing the public to comment).

WORKING TOGETHER TO PROTECT AND RESTORE WETLANDS

In addition to providing regulatory protection for wetlands, EPA works in partnership with states, tribes, and local governments, the private sector, and citizen organizations to monitor, protect, and restore these valuable habitats. EPA is helping states and tribes incorporate wetland monitoring, protection, and restoration into their watershed plans. EPA is also working to develop national guidance on wetland restoration, as well as constructed wetlands used to treat storm water and sewage. Nationally, EPA's Five-Star Restoration Program provides grants and promotes information exchange through community-based education and restoration projects.



If bottomland hardwood swamps are protected, Bald Cypress trees like these can grow for more than 2000 years.



Spotted turtle

Wetland Managers, the National Association of Counties, the Izaak Walton League, local watershed associations, schools, and universities to advance conservation and restoration programs.

HOW CAN I HELP?

You can do many things to help protect wetlands in your area. First, identify your watershed and find the wetlands in your neighborhood. Learn more about them and share what you learn with someone you know! Encourage neighbors, developers, and state and local governments to protect the functions and values of wetlands in your watershed.

To prevent wetland loss or degradation, follow these simple guidelines:

- Invest in wetlands by buying duck stamps. Proceeds from these \$15 migratory bird hunting stamps support wetland acquisition and restoration. The stamps are available on-line at the U.S. Fish and Wildlife Service's web site (www.fws.gov) or at your local post office.
- Rather than draining or filling wetlands, find more compatible uses that would not damage the wetlands, such as waterfowl and wildlife habitat.
- When developing your landscaping plan, keep wetlands in mind. Plant native grasses or forested buffer strips along wetlands on your property to protect water quality.



Wetland habitat along this Idaho riparian corridor provides food and shelter for many different wildlife species

EPA works with a variety of other federal agencies to protect and restore wetlands, including the U.S. Fish and Wildlife Service, the U.S. Department of Agriculture, and the National Marine Fisheries Service. EPA is working with these agencies and others to achieve a net increase of 100,000 acres of wetlands each year by 2005. EPA also partners with private interests and public organizations like the Association of State

- “Get into” your wetland by participating in a volunteer monitoring program.
- Plan to avoid wetlands when developing or improving a site. Get technical assistance from your state environmental agency before you alter a wetland.
- Maintain wetlands and adjacent buffer strips as open space.
- Support your local watershed association.
- Plan a wetland program or invite a wetland expert to speak at your school, club, youth group, or professional organization.
- Build a wetland in your backyard. Learn how by visiting the U.S. Department of Agriculture’s web site at www.nrcs.usda.gov/feature/backyard.

Wetlands can be found in every county and climatic zone in the United States.

CLASSIFICATION OF WETLANDS AND DEEPWATER HABITATS OF THE UNITED STATES

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USE OF THE CLASSIFICATION SYSTEM

This System was designed for use over an extremely wide geographic area and for use by individuals and organizations with varied interests and objectives. The classification employs 5 System names, 8 Subsystem names, 11 Class names, 28 Subclass names, and an unspecified number of Dominance Types. It is, of necessity, a complex System when viewed in its entirety, but use of the System for a specific purpose at a local site should be simple and straightforward. Artificial keys to the Systems and Classes (Appendix E) are furnished to aid the user of the classification, but reference to detailed definitions in the text is also required. The purpose of this section is to illustrate how the System should be used and some of the potential pitfalls that could lead to its misuse.

Before attempting to apply the System, the user should consider four important points:

- (1) Information about the area to be classified must be available before the System can be applied. This information may be in the form of historical data, aerial photographs, brief on-site inspection, or detailed and intensive studies. The System is designed for use at varying degrees of detail. There are few areas for which sufficient information is available to allow the most detailed application of the System. If the level of detail provided by the

data is not sufficient for the needs of the user, additional data gathering is mandatory.

- (2) Below the level of Class, the System is open-ended and incomplete. We give only examples of the vast number of Dominance Types that occur. The user may identify additional Dominance Types and determine where these fit into the classification hierarchy. It is also probable that as the System is used the need for additional Subclasses will become apparent.
- (3) One of the main purposes of the new classification is to ensure uniformity throughout the United States. It is important that the user pay particular attention to the definitions in the classification. Any attempt at modification of these definitions will lead to lack of uniformity in application.
- (4) One of the principal uses of the classification system will be the inventory and mapping of wetlands and deepwater habitats. A classification used in the mapping is scale-specific, both for the minimum size of units mapped and for the degree of detail attainable. It is necessary for the user to develop a specific set of mapping conventions for each application and to demonstrate their relationship to the generalized classification described here. For example, there are a number of possible mapping conventions for a small wetland basin 50 m (164 feet) in diameter with concentric rings of vegetation about the deepest zone. At a scale of 1:500 each zone may be classified and mapped; at 1:20,000 it might be necessary to map the entire basin as one zone and ignore the peripheral bands; and at 1:100,000 the entire wetland basin may be smaller than the smallest mappable unit, and such a small-scale map is seldom adequate for a detailed inventory and must be supplemented by information gathered by sampling. In other areas, it may be necessary to develop mapping conventions for taxa that cannot be easily recognized; for instance, Aquatic Beds in turbid waters may have to be mapped simply as Unconsolidated Bottom.

HIERARCHICAL LEVELS AND MODIFIERS

We have designed the various levels of the system for specific purposes, and the relative importance of each will vary among users. The Systems and Subsystems are most important in applications involving large regions or the entire country. They serve to organize the Classes into meaningful assemblages of information for data storage and retrieval.

The Classes and Subclasses are the most important part of the system for many users and are basic to wetland mapping. Most Classes should be easily recognizable by users in a wide variety of disciplines. However, the Class designations apply to average conditions over a period of years, and since many wetlands are dynamic and subject to rapid changes in appearance, the proper classification of a wetland will frequently require data that span a period of years and several seasons in each of those years.

The Dominance Type is most important to users interested in detailed regional studies. It may be necessary to identify Dominance Types in order to determine which modifying terms are appropriate, because plants and animals present in an area tend to reflect environmental conditions over a period of time. Water regime can be determined from long-term hydrologic studies where these are available. The more common procedure will be to estimate this characteristic from the Dominance Types. Several studies have related water regimes to the presence and distribution of plants or animals (e.g., Stephenson and Stephenson 1972; Stewart and Kantrud 1972; Chapman 1974).

Similarly, we do not intend that salinity measurements be made for all wetlands except where these data are required; often plant species or associations can be used to indicate broad salinity classes. Lists of halophytes have been prepared for both coastal and inland areas (e.g., Duncan 1974; MacDonald and Barbour 1974; Ungar 1974), and a number of floristic and ecological studies have described plants that are indicators of salinity (e.g., Penfound and Hathaway 1938; Moyle 1945; Kurz and Wagner 1957; Dillon 1966; Anderson et al. 1968; Chabreck 1972; Stewart and Kantrud 1972; Ungar 1974).

In areas where the Dominance Types to be expected under different water regimes and types of water chemistry conditions have not been identified, detailed regional studies will be required before the classification can be applied in detail. In areas where detailed soil maps are available, it is also possible to infer water regime and water chemistry from soil series (U.S. Soil Conservation Service, Soil Survey Staff 1975).

Some of the Modifiers are an integral part of this system and their use is essential; others are used only for detailed applications or for special cases. Modifiers are never used with Systems and Subsystems; however, at least one Water Regime Modifier, one Water Chemistry Modifier, and one Soil Modifier must be used at all lower levels in the hierarchy. Use of the Modifiers listed under mixosaline and mixohaline (Table 1) is optional but these finer categories should be used whenever supporting data are available. The user is urged not to rely on single observations of water regime or water chemistry. Such measurements give misleading results in all but the most stable wetlands. If a more detailed Soil Modifier, such as soil order or suborder (U.S. Soil Conservation Service, Soil Survey Staff 1975) can be obtained, it should be used in place of the Modifiers, mineral and organic. Special Modifiers are used where appropriate.

RELATIONSHIP TO OTHER WETLAND CLASSIFICATIONS

There are numerous wetland classifications in use in the United States. Here we relate this system to three published classifications that have gained widespread acceptance. It is not possible to equate these systems directly for several reasons: (1) the criteria selected for establishing categories differ; (2) some of the classifications are not applied consistently in different parts of the country; and (3) the elements classified are not the same in various classifications.

The most widely used classification system in the United States is that of Martin et al. (1953) which was republished in U.S. Fish and Wildlife Service *Circular 39* (Shaw and Fredine 1956). The wetland types are based on criteria such as water depth and permanence, water chemistry, life form of vegetation, and dominant plant species. In Table 4 we compare some of the major components of our system with the type descriptions listed in *Circular 39*.

In response to the need for more detailed wetland classification in the glaciated Northeast, Golet and Larson (1974) refined the freshwater wetland types of *Circular 39* by writing more detailed descriptions and subdividing classes on the basis of finer differences in plant life forms. Golet and Larson's classes are roughly equivalent to Types 1–8 of *Circular 39*, except that they restrict Type 1 to river floodplains. The Golet and Larson system does not recognize the coastal (tidal) fresh wetlands of *Circular 39* (Types 12–14) as a separate category, but classifies these areas in the same manner as nontidal wetlands. In addition to devising 24 subclasses, they also created 5 size categories, 6 site types giving a wetland's hydrologic and topographic location; 8 cover types (modified from Stewart and Kantrud 1971) expressing the distribution and relative proportions of cover and water; 3 vegetative interspersal types; and 6 surrounding habitat types. Since this system is based on the classes of Martin et al. (1953), Table 4 may also be used to compare the Golet and Larson system with the one described here. Although our system does not include size categories and site types, this information will be available from the results of the new inventory of wetlands and deepwater habitats of the United States.

Stewart and Kantrud (1971) devised a new classification system to better serve the needs of researchers and wetland managers in the glaciated prairies. Their system recognizes seven classes of wetlands which are distinguished by the vegetational zone occupying the central or deepest part and covering 5% or more of the wetland basin. The classes thus reflect the wetland's water regime; for example, temporary ponds (Class II) are those where the wet-meadow zone occupies the deepest part of the wetland. Six possible subclasses were created, based on differences in plant species composition that are correlated with variations in average salinity of surface water. The third component of classification in their system is the cover type, which represents differences in the spatial relation of emergent cover to open water or exposed bottom soil. The zones of Stewart and Kantrud's system are readily related to our water regime modifiers (Table 1), and the subclasses are roughly equivalent to our Water Chemistry Modifiers (Fig. 8).

Wetlands represent only one type of land and the classification of this part separate from the rest is done for practical rather than for ecological reasons (Cowardin 1978). Recently there has been a flurry of interest in a holistic approach to land classification (in Land Classification Series, *Journal of Forestry*, vol. 46, no. 10). A number of classifications have been developed (e.g., Radford 1978) or are under development (e.g., Driscoll et al. 1978). Parts

STEWART AND KANTRUD (1972)	APPROXIMATE SPECIFIC CONDUCTANCES (μMhos)	THIS CLASSIFICATION
SALINE	60,000	MIXOSALINE
	45,000	
SUBSALINE	30,000	
	15,000	
BRACKISH	8,000	
	5,000	
MODERATELY BRACKISH	2,000	
SLIGHTLY BRACKISH	800	
	500	
FRESH		FRESH

Figure 1. Comparison of the water chemistry subclasses of Stewart and Kantrud (1972) with Water Chemistry Modifiers used in the present classification system.

Table 1. Comparison of the Zones of Stewart and Kantrud's (1971) Classification with the Water Regime Modifiers used in the Present Classification System

Zone	Water Regime Modifier
Wetland-low-prairie	Non-wetland by our definition
Wet meadow	Temporarily flooded
Shallow marsh	Seasonally flooded
Deep marsh	Semipermanently flooded
	Intermittently exposed
Intermittent-alkali	Intermittently flooded (with eusaline or hypersaline water)
Permanent-open-water	Permanently flooded (with mixohaline water)
Fen (alkaline bog)	Saturated

of this wetland classification can be incorporated into broader hierarchical land classifications.

A classification system is most easily learned through use. To illustrate the application of this system, we have classified a representative group of wetlands and deepwater habitats of the United States (Plates 1–86).

URBAN RUNOFF

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INTRODUCTION

Urban runoff comprises the stormwater runoff from urban areas such as roads and roofs. Road and car park runoff is often more contaminated with heavy metals and organic matter than roof runoff. Runoff is usually collected in gully pots that require regular cleaning. Stormwater pipes transfer urban runoff either directly to watercourses or to sustainable urban drainage systems (also called best

management practice in the United States) such as ponds or wetlands. However, urban runoff is frequently treated only by silt traps that need to be cleaned regularly. This chapter highlights the main areas in urban runoff research and contain general background information supported by case studies.

ROAD RUNOFF DRAINAGE

Background to Road Runoff

Conventional stormwater systems are designed to dispose of rainfall runoff as quickly as possible. This results in 'end of pipe' solutions that often involve large interceptor and relief sewers, huge storage tanks at downstream locations, and centralized wastewater treatment (1–3).

In contrast, sustainable urban drainage systems such as combined attenuation pond and infiltration basin systems (4) can be applied as cost-effective local 'source control' drainage solutions, for example, for collecting road runoff. It is often possible to divert all road runoff for infiltration or storage and subsequent recycling. Runoff from roads is a major contributor to the quantity of surface water requiring disposal, so this is a particularly beneficial approach when suitable ground conditions prevail (1). Furthermore, infiltration of road runoff can reduce the concentration of diffuse pollutants such as oil, leaves, metals, and feces, thereby improving the water quality of surface water runoff (3,4).

Case Study: Design and Operation of Urban Stormwater Ponds Treating Road Runoff

Combined wetlands and infiltration basins are cost-effective 'end of pipe' drainage solutions that can be used for local source control. The aims of this case study were to assess the constraints of the design and operation of these systems, the influence of wetland plants on infiltration rates, and the water treatment potential. Road runoff was first stored and treated in a constructed wetland before it overflowed into parallel infiltration basins; one was planted and the other was unplanted (Fig. 1). British Building Research Establishment and Construction Industry Research and Information Association and German Association for Water, Wastewater and Waste design guidelines were unacceptable because they did not take vertical soil property variations fully into account. Wetland plants in one infiltration basin had no significant influence on drainage properties. The water quality of both infiltration basins was not acceptable for recycling directly after the system setup (5).

ROOF RUNOFF DRAINAGE

Background to Roof Runoff

Combined wet and dry ponds as cost-effective 'end of pipe' drainage solutions can be used for local source control, for example, diversion or collection of roof runoff. It is often possible to divert all roof runoff for infiltration or storage and subsequent recycling. Runoff from roofs is a major contributor to the quantity of surface water requiring



Figure 1. Experimental ponds treating road runoff in Scotland.

disposal, so this is a particularly beneficial approach when suitable ground conditions prevail (1). Furthermore, roof runoff is usually cleaner than road runoff. However, diffuse pollution from the atmosphere and degrading building materials can have a significant impact on the water quality of any surface water runoff (3).

Case Study: Design and Operation of Stormwater Ponds for Roof Runoff Treatment

The purpose of this case study was to optimize design, operation and maintenance guidelines and to assess the water treatment potential of a stormwater pond system after 15 months of operation. The system was based on a combined silt trap, attenuation pond (wet pond) and dry pond construction used for drainage of roof water runoff from a single domestic property (Fig. 2). British Building Research Establishment and Construction Industry Research and Information Association, and German Association for Water, Wastewater and Waste design guidelines were tested. These design guidelines failed because they do not consider local hydrologic and soil conditions. The infiltration function for the dry pond is logarithmic and depends on the season. Furthermore, algae control techniques were successfully applied, and water treatment of rainwater runoff from roofs was largely unnecessary. However, seasonal and diurnal variations in biochemical oxygen demand, dissolved oxygen and pH were recorded (4).

URBAN RUNOFF AND GULLY POT EFFLUENT TREATMENT

Background to Gully Pot Effluent

Gully pots can be viewed as simple physical, chemical and biological reactors. They are particularly effective in retaining suspended solids and settleable solids (6). Currently, gully pot effluent is extracted once or twice per annum from road drains and transported (often over long distances) for treatment at sewage works (1). A more sustainable solution would be to treat gully pot effluent locally in potentially sustainable constructed wetlands,



Figure 2. Experimental infiltration pond for roof runoff in England.

reducing transport and treatment costs. Furthermore, gully pot effluent treated by constructed wetlands can be recycled for cleansing gully pots and washing wastewater tankers. Heavy metals within urban runoff originate from fuel additives, car body corrosion, and tire and brake wear. Common metal pollutants from road vehicles include lead, zinc, copper, chromium, nickel and cadmium. Typical concentration ranges for most heavy metals are between 0.05 and 2 mg/L (7).

Case Study: Design and Operation of Constructed Wetlands Applied to Gully Pot Effluent Contaminated with Heavy Metals

The aim was to assess the treatment efficiencies for gully pot effluent of experimental vertical-flow constructed wetland filters containing *Phragmites australis* (Cav.) Trin. ex Steud. (common reed) and filter media of different adsorption capacities. Six out of 12 filters received inflow water spiked with heavy metals. For one year, hydrated copper nitrate and hydrated nickel nitrate were added to sieved gully pot effluent to simulate contaminated primary treated stormwater runoff. The inflow concentrations for dissolved copper, nickel and nitrate were approximately 1.0, 1.0 and 3.7 mg/L, respectively. For those six filters receiving metals, an obvious breakthrough of dissolved nickel was recorded after road gritting and salting during winter. Sodium chloride was responsible for leaching of dissolved nickel (Fig. 3). Reductions of copper, nickel, biochemical oxygen demand, and suspended solids were frequently insufficient compared to international secondary wastewater treatment standards. An analysis of variance indicated that all filters were similar in their treatment performance (8).

URBAN RUNOFF DISCHARGING INTO URBAN WATERCOURSES

Background to Urban Runoff Discharge to Waters

Stormwater runoff is usually transferred within a combined sewer (rainwater and sewage) or separate stormwater pipeline. Depending on the degree of pollution,

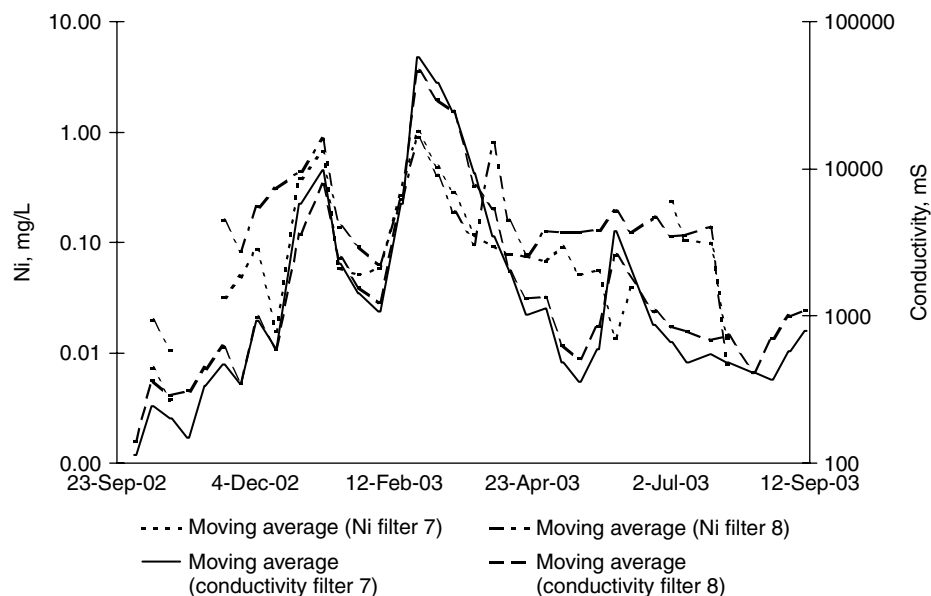


Figure 3. Relationship between the breakthrough of nickel (Ni) and conductivity within the outflows of two experimental constructed wetlands.

it may require further conventional treatment at the wastewater treatment plant. Alternatively, stormwater runoff may be treated by more sustainable technologies, including stormwater ponds, or simply by disposing it via ground infiltration or drainage into a local watercourse (1).

Common preliminary treatment steps for stormwater that is to be discharged into a local watercourse are extended storage within the stormwater pipeline and transfer through a silt trap (also referred to as a sediment trap). The silt trap is often the only active preliminary treatment of surface water runoff despite the fact that stormwater is frequently contaminated with heavy metals and hydrocarbons (9). Heavy metals within road runoff are from fuel additives, car body corrosion, and tire and brake wear (1).

The dimensioning of silt traps is based predominantly on the settlement properties of the particles and the maximum flow-through velocity. As a general rule, the outflow should be located on the same axis as the inflow, so that continuous flow ensures particle settlement over the full length of the silt trap (10). The structure should be small and simple to keep capital costs low.

Various international and British guidelines exist for designing outflow structures. In practice, designs vary considerably. However, silt traps are usually longer than wider by a factor of up to five. Silt trap depths vary considerably but are usually about 40 cm. However, often the lack of maintenance of outflow structures such as silt traps leads to problems years after the structure has been commissioned. Accumulated sediment has to be removed from silt traps to retain their original design performance and to avoid resuspension of accumulated pollutants during storms, causing pollution in receiving waterbodies (11). Watercourses in urban (built-up) areas that receive surface water runoff may be subject to pollution, particularly, if silt traps, for example, are insufficiently maintained (e.g., low frequency of sediment removal). Furthermore, the additional water load may contribute to local flooding (11).

Case Study: Urban Runoff Quality Associated with a Full Silt Trap Discharging into an Urban Watercourse

The aim was to assess the influence of a full silt trap at the end of a stormwater drainage pipe on the water quality of stormwater discharged into a semi-natural urban watercourse. For approximately eleven weeks, the water qualities of the preliminarily treated stormwater and of the receiving watercourse (Braid Burn) were studied. The mean outflow concentrations of suspended solids were 2.0 and 34.1 mg/L during dry and wet weather conditions, respectively. Suspended solids concentrations of up to 141.6 mg/L were recorded during storms. Suspended solids values for treated stormwater were often too high compared to international secondary wastewater treatment standards of approximately 30 mg/L. Pollutants, including heavy metals (e.g., zinc, copper, and nickel) accumulated in the silt trap (Fig. 4). However, high outflow velocities during heavy rainfalls did not result in clearly defined sediment layers due to sediment resuspension. Metals did not accumulate in the receiving watercourse (12).

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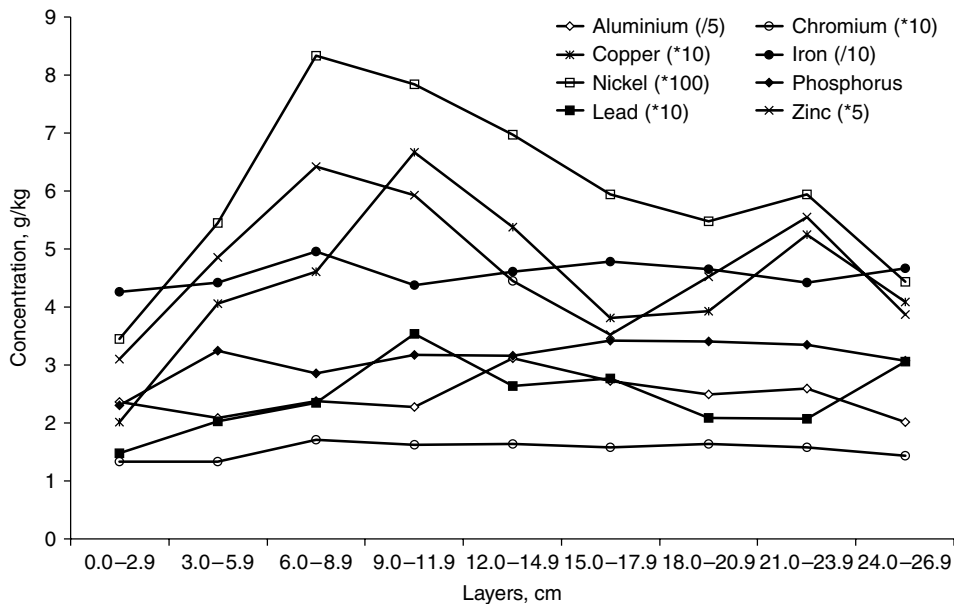


Figure 4. Metal concentrations in different sediment layers of a silt trap in Scotland.

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URBAN WATER STUDIES

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INTRODUCTION

The urban landscape contains many elements that can alter the water quality of a waterbody. Before development occurs, grass and other natural vegetation hold sediment in place. Streams flowing through undeveloped terrain have no major sources of contaminants, such as bacteria

(except for wildlife), that affect contact recreation. There are also no major sources of toxins that could negatively impact humans and wildlife. Development (residential, commercial, industrial, and agricultural, for example) brings issues that have the potential to compromise the character of streams through point source pollution or nonpoint source pollution (Fig. 1). The best way to prevent or deal with these issues is to have cooperation among a broad group of stakeholders in each watershed, the area of land (and tributaries) that drains to a particular waterbody. These individuals must work together to develop sound watershed management to protect or to restore the health of the ecosystem of that watershed.

WHAT HAPPENS TO THE QUALITY OF WATER WHEN DEVELOPMENT BEGINS?

One of the first concerns for water quality during development involves construction sites. When the natural vegetation is removed and bare soil is exposed, rainwater easily collects sediment, along with any pollutant attached to that sediment, and drains eventually to a stream. Additional sediment added to the water column may pose impacts to humans, such as complications with sources of public water supply, as well as to aquatic organisms. Sediment reduces sunlight for aquatic photosynthesis, clogs fish gills, disturbs bottom-dwelling organisms, and overall reduces the aesthetic value of the stream.

DURING AND AFTER DEVELOPMENT—POINT SOURCE AND NONPOINT SOURCE POLLUTION

Development includes the establishment of industrial plants, wastewater treatment plants, and other entities that discharge treated water to an outfall that connects to a stream. State and/or federal laws and standards regulate the effluent (point source) of these plants, but there still lies the potential for overflows, illegal discharges, and accidental spills.

Although there have been major strides in controlling pollutants from point sources, nonpoint source pollution (NPS) remains one of the most complex and elusive problems to deal with in watershed management. NPS pollution stems from those constituents that flow overground from diffuse sources directly into area waterways or indirectly through stormwater infrastructure to outfalls that connect to rivers and streams. One important factor to remember is that water that flows from homes to wastewater treatment facilities and then to a specified waterbody is clean as it leaves the plant. Stormwater receives no treatment. Left uncontrolled, these pollutants often have a significant impact on the quality of the water.

Some potential NPS pollutants carried in rainwater runoff include the following:

- excessive fertilizer applied to lawns or agricultural fields
- excessive pesticides and herbicides applied to lawns or agricultural fields
- grass clippings left on streets or blown down storm drains
- motor oil or other vehicle fluids from leaking automobiles or from illegal dumping
- trash
- animal waste
- sewage from broken sewer lines
- sewage from failing septic systems

Some of the consequences of these components entering a waterway include the following:

- excessive nutrient levels that may lead to eutrophic conditions
- depressed dissolved oxygen levels that can be harmful to aquatic species
- harmful elements exposed to wildlife
- increased bacteria counts
- impaired aesthetics
- impaired recreational value

APPROACHING THE ISSUES—WATERSHED MANAGEMENT

When analyzing water quality for an urban area, a watershed approach is the most effective means of defining problem areas and addressing concerns. Studying the various aspects within the watershed and how they interact is a scientifically sound approach to preventing and to solving water quality problems. Some of the various layers of information easily available to use in conjunction with water quality data include topographic information, detailed drainage information, soil types and characteristics, groundwater/aquifer information, vegetation types, climate information, and geology. Land cover or land use data can be obtained from aerial photography or other digital imagery.

After gathering this type of information, participants in the watershed management plan may locate individual potential pollution sources, such as treatment plant



Figure 1. An urban stream with many potential sources of pollution.

or stormwater outfall locations. They may also locate potential areas of concern, such as areas of concentrated industrial activity, areas with large numbers of septic tanks, or areas with concentrated animal feeding operations. The latter activities are commonly found in rural areas, but these operations in the watershed influence the character of the waterbody as well.

Gathering of individuals, or stakeholders, from agriculture, industry, local business, government, local neighborhoods, the environmental profession, as well as other interested parties is the next step in addressing issues on a watershed scale. Each of these individuals can lend support in various aspects of the activities within the watershed.

Monitoring

Due to the size of urban areas, there are typically a number of entities that measure water quality for specific purposes. Gathering this information and coordinating among different groups, including city offices, county offices, river authorities, and other environmental groups can lead to a wealth of information about water quality throughout a watershed. Many agencies may be willing to modify their programs to help collect data in certain areas that are not monitored by any other entity. This action is commonly referred to as coordinated monitoring.

Common parameters used to assess the overall health of a waterbody include the following:

Field Measurements

- dissolved oxygen
- pH
- water temperature
- specific conductance

- salinity in marine waters
- Secchi depth
- flow
- days since last significant rainfall
- observational information, including water appearance and weather (and as applicable, biological activity; pertinent observations related to water quality or stream uses, e.g., boating, swimming, fishing, irrigation pumps, etc.; watershed or in-stream activities, e.g., bridge construction; unusual odors; specific sample information; and missing parameters)

Conventional Chemical Parameters

- ammonia –N
- nitrate + nitrite –N
- total phosphorus
- orthophosphorus
- chlorophyll *a*
- pheophytin
- total dissolved solids
- total suspended solids
- sulfate
- chloride
- total hardness

Bacteria

- fecal coliform
- *E. Coli* or *Enterococcus*

Flow Measurement

- direct measurement or USGS gage data
- flow estimate

Additional parameters for total maximum daily load (TMDL) purposes, load calculations, modeling, trend analysis, etc. or other specific purposes not related to the 305(b) screening:

- alkalinity
- total Kjeldahl nitrogen
- total organic carbon
- BOD5
- COD
- Toxics (metals or organics in water) for aquatic life/habitat monitoring

Routine monitoring should not include ambient toxicity, toxics in sediment or fish tissue, 24-hour dissolved oxygen measurements, and monitoring to determine the extent of impairment.

In addition to collecting data on water chemistry, biological and habitat data can assist in more comprehensively understanding the health of a watershed.

DEFINING ISSUES

Approaching concerns at a watershed level is appropriate in most situations; however, there may be issues confined to only one part of the watershed. Based on available

data, researchers may be able to pinpoint a problem to a certain area. For example, the main stem of a channel may be contaminated with bacteria from a certain point downstream of a confluence with a tributary. Detailed study of that tributary's subwatershed may lead to sources of the bacteria that may help resolve the overall issue. So, while taking a watershed approach to water quality, it sometimes may be beneficial to go a step further to the subwatershed level, especially when trying to mitigate a particular pollutant. Eliminating or simply reducing input from small tributaries is the first step in cleaning up the entire watershed.

PREVENTING THE PROBLEMS—BEST MANAGEMENT PRACTICES AND PUBLIC EDUCATION

An important component of addressing water quality problems associated with NPS is through a preventive approach with using best management practices (BMP) at construction sites and other facilities to prevent contaminated runoff. Another strong preventive is public education. Once people know they can have a positive impact on the waterbodies in their community and watershed, then communities are one step closer to establishing healthy ecosystems in urban areas.

CONCLUSION

Urban areas often contain an immense network of flowing waterbodies containing quality aquatic habitat and recreational opportunities such as boating, canoeing, fishing, and swimming. At the same time, the many waterbodies that run through highly urbanized or impacted landscapes face challenges that require the cooperation of various entities to produce sound watershed management. The maintenance of a network of stakeholders and monitoring entities that works toward this goal is essential. Preventive steps such as BMPs, public education, and outreach opportunities are also very important in creating and maintaining quality places in which to live.

SUBGLACIAL LAKE VOSTOK

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When Captain Robert F. Scott first observed the McMurdo Dry Valleys in December 1903, he wrote, "We have seen no living thing, not even a moss or a lichen. . . it is certainly a valley of the dead." Eight years later, Scott's team reached the South Pole and his entry referencing conditions on the polar plateau read "Great God! This is an awful place." Scott was unaware that life surrounded him in the dry valleys, and he could not have realized that microbial life could exist miles beneath his feet in an environment sealed from the atmosphere by Antarctica's expansive continental ice sheet. The realization that there was life

on the Antarctic continent, other than that associated with the marine system, did not come to light until the seminal investigations initiated by the International Geophysical Year in the late 1950s and early 1960s. Biological studies during this period were exploratory in nature, describing life forms and their habitat in terrestrial and surface lake environments. These important pioneering studies reshaped our understanding of the potential for life in the coldest, driest, and windiest place on Earth. However, it was not until about 5 years ago that microbiological evidence suggested that the Antarctic interior could serve as an environment for life, which resulted from investigations conducted on a deep ice core from Vostok Station. These studies suggested that the cryosphere of Antarctica could support some of the most unusual and extreme microbial ecosystems on the planet. This information, coupled with the recent discovery of an extensive network of subglacial lakes beneath the East

Antarctic Ice Sheet (EAIS), has changed our view of life in Antarctica. Although it is well documented and widely known that life exists in the permanently cold margins and marine environments of Antarctica, the cryosphere has traditionally been viewed as being devoid of life.

Subglacial Lake Vostok is by far the largest of the more than 140 subglacial lakes that have been identified thus far (Fig. 1), with a surface area of more than 14,000 km² and a depth in excess of 800 m, which makes it one of the largest lakes on Earth. The base of the central EAIS is relatively warm as a result of the combined effect of geothermal heating and the insulation caused by the overlying ~4 km of ice, despite surface air temperatures commonly below –60 °C. Lake Vostok has only recently become a focus of research, even though its presence was first noted in the early 1960s when Soviet Antarctic Expedition (SAE) pilots observed an extremely flat area near Vostok Station that could represent ice floating over water. These airborne

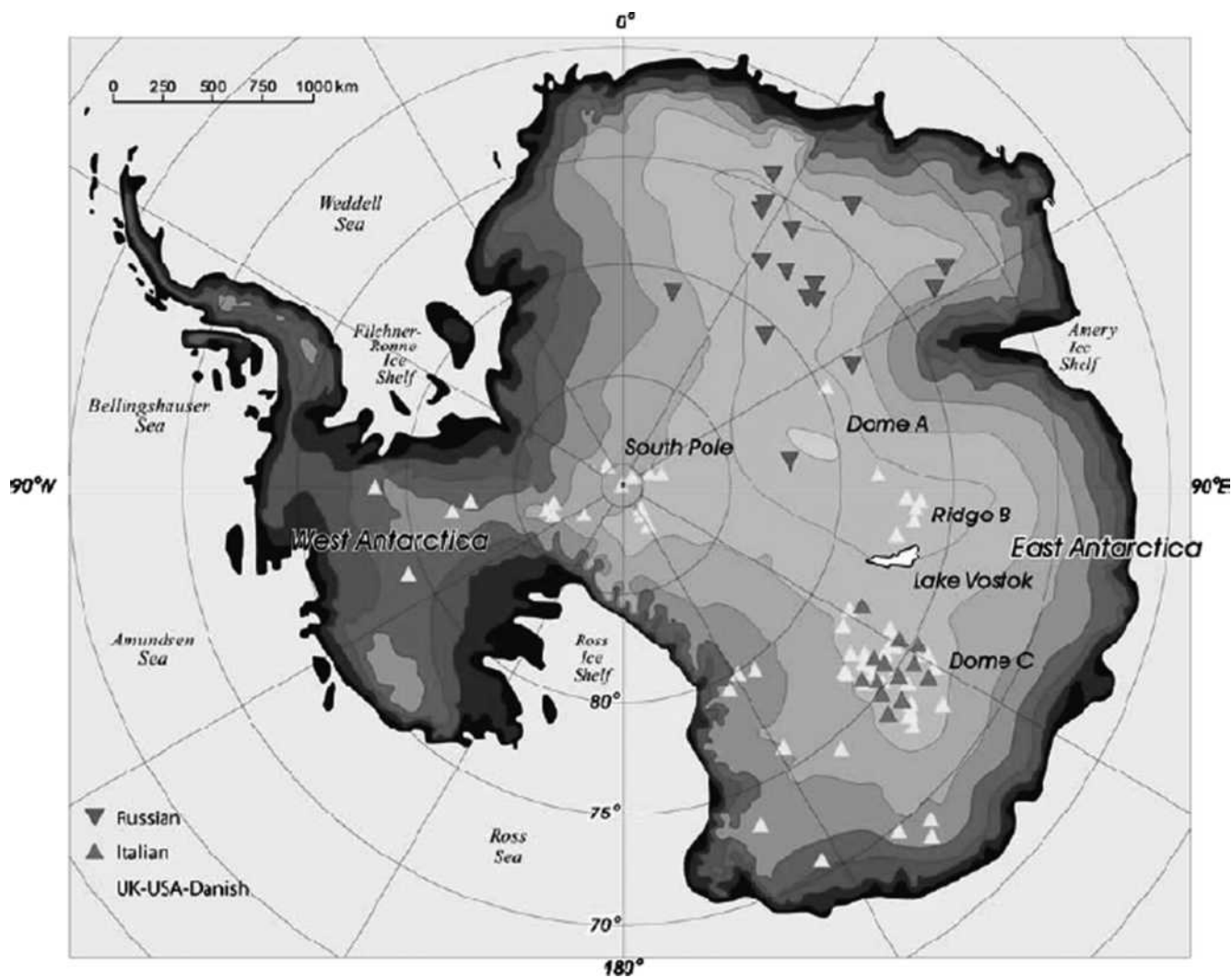


Figure 1. Locations of Antarctic subglacial lakes. Lakes discovered by Italian (blue triangles), Russian (red down white triangles), and U.K.–U.S.–Danish (yellow triangles) teams are included (Lake Vostok is shown in outline). The ice-sheet surface is contoured at 500-m intervals. Reproduced from Priscu et al., 'Subglacial Antarctic Lake Environments: International Planning for Exploration and Research.' EOS, in press.

observations were soon confirmed by seismic data, radar profiles, and satellite images made near Vostok Station. The discovery of Lake Vostok and several additional subglacial lakes in central Antarctica during the early 1970s went relatively unnoticed by the biological scientific community until the late 1990s. However, curiosity about the nature of this environment and the potential existence of microbial life in subglacial lakes has recently intensified.

The site selection for Vostok Station by the SAE was truly fortuitous for subglacial lake research. Several deep ice cores have been extracted from the ice sheet at Vostok

Station since drilling began in the mid-1960s (the first 500-m-deep borehole was made in 1965). The most recent and deepest (3623 m below the surface) ice core (borehole 5G) stopped ~120 m above the lake-ice interface, because of concerns of contaminating the lake. The upper 3310 m of the ice core has provided a detailed paleoclimate record spanning the past 420,000 years. Ice at depths between 3310 and 3538 m is deformed by contact with bedrock and does not contain useful paleoclimate data. The basal portion of the ice core from 3539 to 3623 m has a chemistry and crystallography distinctly different from the “normal”

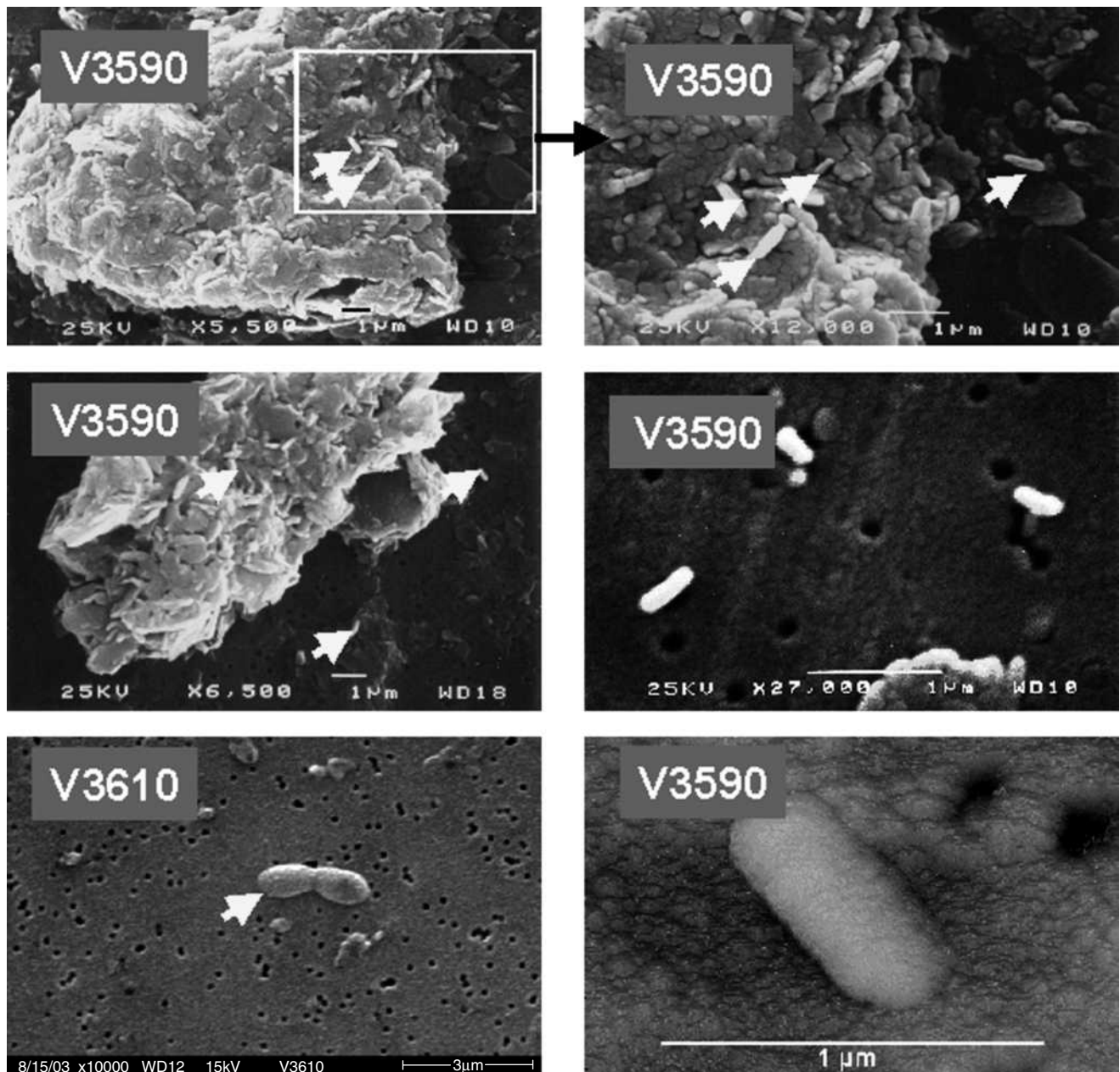


Figure 2. Scanning electron micrographs of microbes in Lake Vostok accretion ice. Samples are designated by their depth of recovery from below the surface in meters, and yellow arrows indicate prokaryotic cells. Melt water was filtered onto 0.2- μ m polycarbonate filters sputter coated with 10-nm Au-Pd and imaged with a cryogenic scanning electron microscope. Adapted from Priscu et al., *Science* **286**: 2141–2144 (1999).

glacial ice above it. This basal ice has extremely low conductivity, large (up to 1 m) ice crystals, and numerous sediment inclusions. The mineral composition of ice-bound sediments below 3539 m is dominated by micas and is clearly different than typical crustal composition and particles within the overlying glacial ice. The isotopic composition of this basal ice in concert with other data indicates that it formed by the refreezing (accretion) of Lake Vostok lake water to the underside of the ice sheet. Thus, there is ~210 m of Lake Vostok water frozen on the underside of the glacial ice beneath Vostok Station. Recent airborne radar surveys have shown that the accretion ice is dynamic and actually flows out of the lake on the downstream side.

The freshwater in Lake Vostok originates from the overlying ice sheet, which melts near the shoreline of the lake and at the ice–water interface in the north. Accretion to the base of the ice sheet occurs in the central and southern regions, removing water from the lake. As the accreted ice is essentially gas-free, the lake water is thought to have a relatively high dissolved gas content supplied from air bubbles released from the overlying glacial ice. Estimates on the gas content of Lake Vostok suggest that it could have an oxygen concentration 50 times higher than that in the open ocean. Although it seems inevitable that viable microorganisms from the overlying glacial ice and in sediment scoured from bedrock adjacent to the lake are regularly seeded into the lake, the question remains whether these or preexisting microorganisms have established a flourishing community within Lake Vostok. Evidence for the existence of geothermal activity is supported by the recent interpretation by French scientists of He^3/He^4 data from accretion ice. These data imply that there may be extensive faulting beneath Lake Vostok, which results in geothermal plumes entering the bottom of the lake in the southern part of Lake Vostok. If this emerging picture is correct, Lake Vostok could harbor an ecosystem fueled by geochemical energy much like that observed in deep-sea vent systems.

Because of concerns of contaminating these pristine subglacial environments and the current lack of an adequate sampling strategy to meet these needs, no direct samples have yet been obtained from subglacial lakes. Therefore, the accreted ice retrieved from Vostok Station has offered the only opportunity for microbial ecologists and geochemists to investigate material derived from a subglacial lake. Microbiological and molecular-based studies of accreted ice by four independent laboratories have indicated that these cores contain bacteria (Fig. 2). Metabolic and culturing experiments demonstrate that a portion of the assemblage becomes metabolically active upon melting and exposure to nutrients which suggests Lake Vostok contains a viable microbial community. Molecular profiling of microbes within the accreted ice (by both culturing and direct 16S rDNA amplification) shows close agreement with present-day surface microbiota, consisting of phylotypes within the bacterial divisions *Proteobacteria* (alpha, beta, gamma, and delta), Low G + C Gram Positives, *Actinobacteria*, and the *Cytophaga*–*Flavobacterium*–*Bacteroides* line of descent.

If microbes within the accreted ice are representative of the lake microbiota, this would imply that bacteria within Lake Vostok do not represent an evolutionarily distinct subglacial biota. The time scale of isolation within Lake Vostok (~20–25 million years) is not long in terms of prokaryotic evolution compared with their 3.7×10^9 year history on Earth. An alternative scenario is that glacial melt water entering the lake forms a lens overlying the Lake Vostok water column. If so, the microbes entrapped within accretion ice would likely have spent little time within the actual lake water (few, if any cell divisions occurring) before being frozen within the accretion ice. The microbes within the main body of the lake below such a freshwater lens may have originated primarily from basal sediments, rocks, and/or the preexisting environment before Lake Vostok became ice-covered. If so, their period of isolation may be adequate for significant evolutionary divergence, particularly given the potential selection pressures that may exist within this unusual subglacial environment.

Epifluorescence (of DNA-stained cells) and scanning electron microscopy have measured bacterial cell densities of 2.8×10^3 and 3.6×10^4 cells mL^{-1} , respectively, in Vostok accretion ice (3590 m below the surface). Additionally, the concentration of dissolved organic carbon (DOC) in this core was measured at 0.51 mg L^{-1} . Based on these values and partitioning coefficients for the water to ice phase change, it is estimated that the water in Lake Vostok had bacterial cell concentrations of 10^5 to 10^6 mL^{-1} and a DOC concentration of 1.2 mg L^{-1} . Using data on the volume of the Antarctic ice sheet and subglacial lakes in concert with published bacterial volume to carbon conversion factors, the cell number and carbon content within the Antarctic ice sheet and subglacial lakes can be estimated, assuming that concentrations of bacterial cells and DOC in other subglacial lakes are similar to those estimated for Lake Vostok. Based on these calculations, subglacial lakes contain about 12% of the total cell number and cell carbon with respect to the pools associated with the Antarctic continent (e.g., subglacial lakes + ice sheet). The number of cells and prokaryotic cell carbon content for subglacial lakes plus the ice sheet (1.00×10^{26} cells; 2.77×10^{-3} Pg) approaches that reported for Earth's freshwater lakes and rivers (1.3×10^{26} ; 2.99×10^{-3} Pg), which suggests these environments contain nontrivial amounts of cells and carbon. These estimates of the number of prokaryotes and organic carbon associated with Antarctic subglacial lakes and glacial ice are clearly tentative and should be refined once additional data become available; however, they do imply that Antarctica contains an organic carbon reservoir that should be considered when addressing issues concerning global total carbon storage reservoirs and dynamics.

The seminal reports of life in subglacial Antarctic lakes have spawned many new ideas about how life evolved on our planet and others, how life adapts to extremely cold conditions, and how ecosystems function beneath 4 km of ice, and it has changed our view of the extent of Earth's biosphere. One new idea theorizes that microbes might thrive throughout most of the Antarctic ice sheet within intercrystalline veins. The veins, formed at junctions

where three ice crystals meet, may offer a unique habitat for microbes in “solid” ice that has not yet been considered. These veins contain a relatively solute-rich solution that both lowers the freezing point and provides nutrition for microbial growth. This idea is supported by new data on the isotopic composition of gases within cores of Vostok glacial ice, which suggests biological modification. If it is eventually shown that such a microbial habitat does indeed exist in solid ice, then much of the Antarctic ice sheet may harbor active microbes, a notion previously unheard of. Important new information based on airborne radar surveys has shown that at least 140 more subglacial lakes exist beneath the EAIS. Most of these lakes exist near Dome C. These lakes, although not nearly as large as Vostok, are still substantial with at least one exceeding 500 km² in surface area. The density of lakes near Dome C, together with maps of bottom topography, implies that a hydrologically interconnected lake district may exist in this region of Antarctica. These lakes provide many new environments that presumably contain life in what was once thought to be an inhospitable environment.

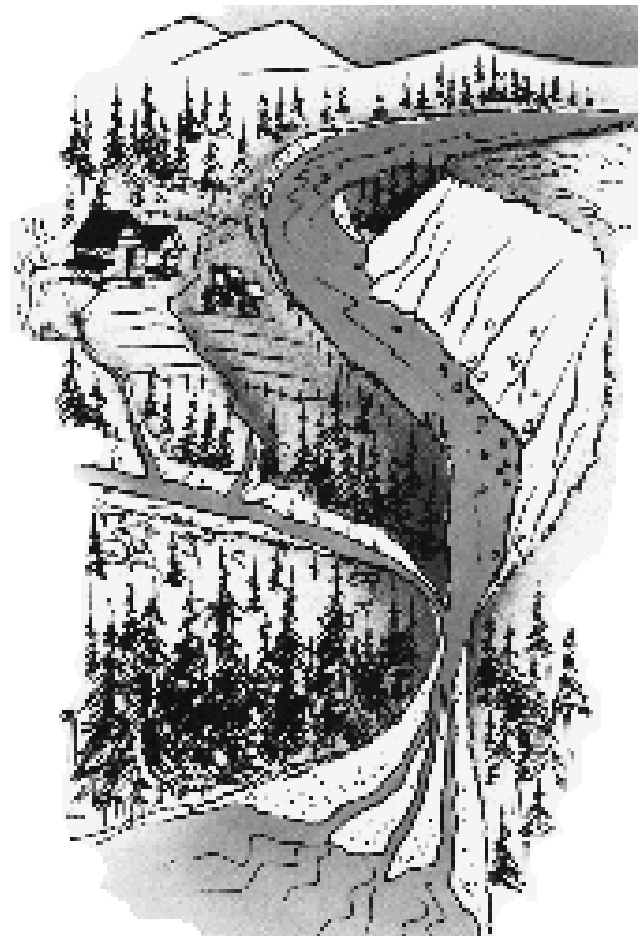
Future research of microbial ecosystems in subglacial lakes depends on a plan in which water samples are collected and returned to the surface. The overriding and limiting issues of this entire strategy are environmental concerns and the control of contamination in both forward and return excursions into the lakes. As such, it is of prime importance that environmental stewardship precedes all scientific endeavors, and that such an undertaking not be attempted until adequate sampling techniques are established. To this end, the Scientific Committee on Antarctic Research (SCAR) has established an international body of specialists to outline a detailed plan for eventual lake entry and sample return. This plan calls for the establishment of a network of instruments that gather limnological data continuously, collection of water samples for return to the surface, and recovery of deep sediment cores that can reconstruct paleoclimate and geological records for Antarctica. The next 10 years should prove to be an interesting time of discovery for Antarctic science, one that follows the Antarctic tradition of melding interdisciplinary and international science. We can expect subglacial lakes to be at the forefront of such discovery because they remain one of the last unexplored frontiers on our planet.

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WATER—THE CANADIAN TRANSPORTER

Environment Canada



Water plays an important role in the transformation of the Canadian landscape by moving large amounts of soil, in the form of *sediment*. Sediment is eroded from the landscape, transported by river systems, and eventually deposited in a lake or the sea. For example, the Fraser River carries an average of 20 million tonnes of sediment a year into the marine environment.

The sediment cycle starts with the process of *erosion*, whereby particles or fragments are weathered from rock

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material. Action by water, wind, glaciers, and plant and animal activities all contribute to the erosion of the earth's surface. Fluvial sediment is the term used to describe the case where water is the key agent for erosion. Natural, or geologic, erosion takes place slowly, over centuries or millennia. Erosion that occurs as a result of human activity may take place much faster. It is important to understand the role of each when studying sediment transport.

Any material that can be dislodged is ready to be transported. The *transportation* process is initiated on the land surface when raindrops result in sheet erosion. Rills, gullies, streams, and rivers then act as conduits for sediment movement. The greater the discharge, or rate of flow, the higher the capacity there is for sediment transport.

The final process in the cycle is *deposition*. When there is not enough energy to transport the sediment, it comes to rest. Sinks, or depositional areas, can be visible as newly deposited material on a flood plain, bars and islands in a channel, and deltas. Considerable deposition occurs that may not be apparent, as on lake and river beds. A knowledge of sediment dynamics is an integral part of understanding the aquatic ecosystem.

Canada's waterways move many millions of tonnes of sediment annually in this never-ending cycle of erosion, transportation, and deposition. Sediment is measured and classified according to its dynamic characteristics:

- suspended load (suspended in the water)
- bed load (rolling or bouncing along the bottom)
- bed material (stationary on the bed)

WHY IS SEDIMENT IMPORTANT?

Sediment carried in water has a variety of effects: what are they and why are they important?

Toxic Chemicals

Sediment plays a major role in the transport and fate of pollutants and so is clearly a concern in water quality management. Toxic chemicals can become attached, or adsorbed, to sediment particles and then transported to and deposited in other areas. These pollutants may later be released into the environment. By studying the quantity, quality, and characteristics of sediment in the stream, scientists and engineers can determine the sources and evaluate the impact of the pollutants on the aquatic environment. Once the sources and impact are known, action can be taken to reduce the pollutants. The association of toxic chemicals with sediment is an issue of national importance.

Navigation

Deposition of sediment in rivers or lakes can decrease water depth, making navigation difficult or impossible. To ensure access, some of the sediment may be dredged from the stream or harbour, but this may release toxic chemicals into the environment. To determine how much dredging needs to be done and how often, water levels

must be monitored, and the rates of sediment transport and deposition estimated. Sedimentation of navigation channels is a concern in the Fraser River (British Columbia), the Mackenzie River (Northwest Territories), and the Great Lakes-St. Lawrence system (Ontario and Quebec).

Fisheries/Aquatic Habitat

Streamborne sediment directly affects fish populations in several ways:

- Suspended sediment decreases the penetration of light into the water. This affects fish feeding and schooling practices, and can lead to reduced survival.
- Suspended sediment in high concentrations irritates the gills of fish, and can cause death.
- Sediment can destroy the protective mucous covering the eyes and scales of fish, making them more susceptible to infection and disease.
- Sediment particles absorb warmth from the sun and thus increase water temperature. This can stress some species of fish.
- Suspended sediment in high concentrations can dislodge plants, invertebrates, and insects in the stream bed. This affects the food source of fish, and can result in smaller and fewer fish.
- Settling sediments can bury and suffocate fish eggs.
- Sediment particles can carry toxic agricultural and industrial compounds. If these are released in the habitat they can cause abnormalities or death in the fish.

Forestry

Some forestry practices have negative impacts on the environment. Extensive tree cutting in an area may not only destroy habitat but increase natural water runoff and accelerate soil erosion. These can lead to increased flow and sediment loads in nearby streams. They can also release chemical substances occurring naturally in forest soils, and allow them to contaminate rivers or lakes. Both the chemicals and the additional sediment can harm fish and other organisms. Sediment problems resulting from forestry practices are prevalent in British Columbia, Ontario, Quebec, New Brunswick, and Newfoundland.

Water Supply

Sediment can affect the delivery of water. When water is taken from streams and lakes for domestic, industrial, and agricultural uses, the presence of sediment in the water can wear out the pumps and turbines. As this increases maintenance costs, it is important to determine the amount of sediment in the stream so that the appropriate equipment can be chosen when designing a water supply plant.

Energy Production

The amount of sediment transported affects both the size and the life expectancy of reservoirs created for power

generation. A dam traps sediment that would normally be carried downstream, and that sediment decreases the size of the reservoir and thus its use for power generation. Therefore, it is necessary to know the amount of sediment to ensure the effective design of reservoirs for the long term.

Agriculture

Some farming practices increase soil erosion and add toxic chemicals to the environment. Thus, productive soil is lost to farms, sediment and pollutants are added to streams, and maintenance costs of irrigation systems are increased. Sediment data and information are necessary in the evaluation of cropping practices and their environmental effects. Sediment-related problems associated with agriculture occur across the country.

Case Study: Aquatic Habitat and Construction In St. John's, Newfoundland

A considerable amount of sediment is lost from building sites during construction. Depending on its location, this sediment finds its way to sewer or stream systems, increasing the costs of water treatment or impacting on aquatic habitat. In 1982, construction was started on a building to house the Institute for Marine Dynamics in St. John's, Newfoundland. The site, near the Rennie's River, required massive excavation. To minimize the impact of the sediment on the aquatic environment, a de-sedimentation facility was built. A settling pond was used, and alum was added to help settle the sediment, alum having no significant impact on the ecosystem. Without the de-sedimentation facility, toxic chemicals and sediment could have contaminated the fish, buried fish eggs, dislodged aquatic plants, and generally overwhelmed the aquatic habitat. Over three years, 1250 tonnes of sediment were kept from entering the river waters. The cost of the de-sedimentation was less than one tenth of one percent of the building costs, and a highly productive trout habitat was protected from contamination.

HOW IS SEDIMENT SAMPLED?

Sediment quantity and quality are sampled in a variety of ways:

- Specially designed *suspended-sediment* samplers are used to collect water/sediment samples that are analyzed for sediment quantity and sometimes quality.
- *Bed-load* samples are usually taken by lowering a specially made sampler to the stream bed. Resting there, the sampler traps the material moving along the bottom.
- *Bed-material* samples may be taken simply by hand from exposed bars or stream banks, or by samplers from the stream bed. Some samplers scoop sediment by simply digging into the bed, while other kinds extract a core from the bed.

Once collected, suspended-sediment samples are analyzed for concentration and particle size. This is usually done in a laboratory. The *concentration* is the ratio of sediment (dry weight) to the total water-sediment mixture, expressed as milligrams per litre (mg/L). The *particle size* is simply the size of the sediment particles. Depending on their size, they are classified as sand, silt, or clay.

To find out how much material is transported by a river, one can combine the concentration with the stream discharge, or flow. This gives the sediment load, which indicates the total amount of sediment transported over a certain time period, whether an hour, a day, month, or year. In such a way, it has been estimated that at Montreal the St. Lawrence River transports 2.3 million tonnes of sediment in suspension each year, or the equivalent of 230,000 truck-loads of soil.

Water, and more specifically the hydrologic cycle, plays a major role in driving the sediment life cycle. The amount of water and its distribution over time influence how and when sediment is sampled.

In Canada, sampling is usually for suspended-sediment data. Most of this sampling is done during high-flow conditions (spring, summer, and fall rainstorms), when most of the sediment is transported through the river system. However, a few samples may be taken at other times throughout the year, to better define the sediment regime. Bed-load sampling is typically undertaken in the spring, when high discharge mobilizes the stream bed. Bed-material may be sampled during the summer, when low-flow conditions may expose parts of the stream bed, making sampling easier.

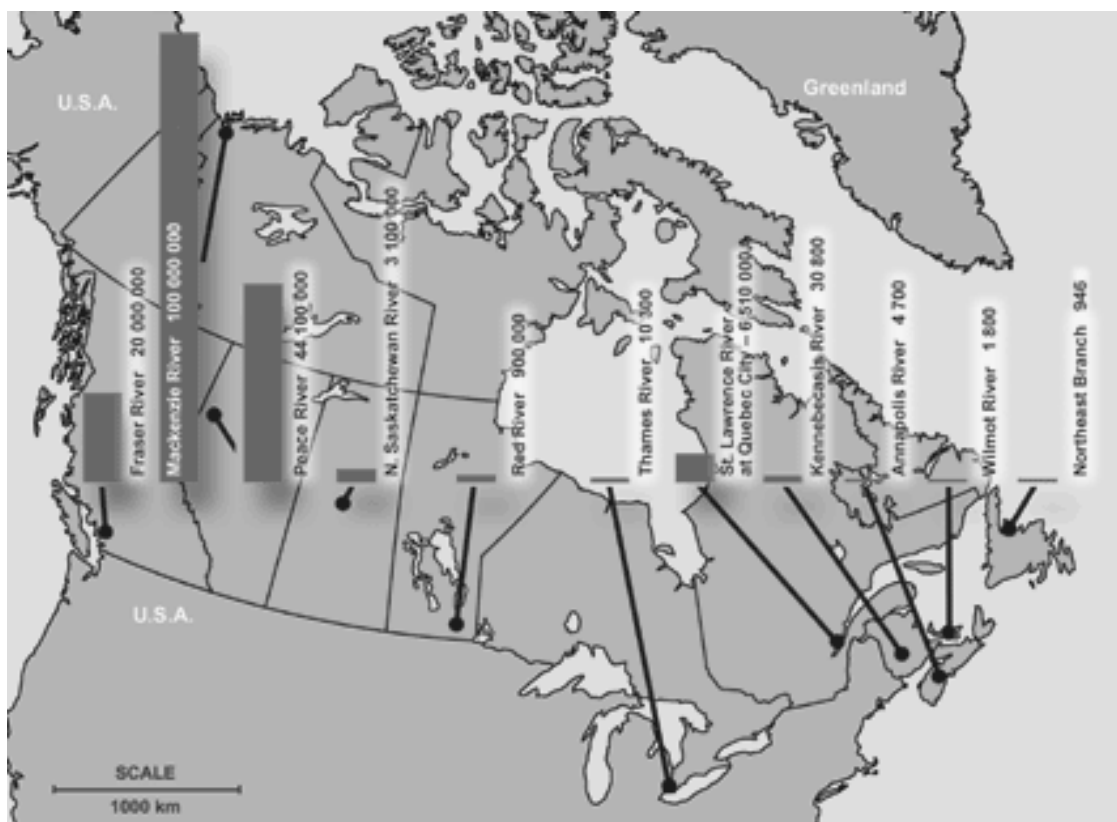
Once the samples have been analyzed, the data on concentrations, particle sizes, or loads can be applied to engineering and environmental questions.

Average Annual Suspended Sediment Load (tonnes) for Selected Rivers in Canada

Glaciers retreated 10,000 years ago, leaving large amounts of easily erodible material across much of western Canada. In mountainous areas (e.g., the Fraser, Peace, and upper Mackenzie rivers), steepness and abundant water supply enable large amounts of sediment to be carried away. In contrast, the flat and dry conditions of the Prairies result in much lower sediment loads. In eastern Canada, where much of the land is bedrock, there is a limited sediment supply and therefore smaller loads.

SEDIMENT DATA AND INFORMATION

The measurement of sediments in streams in Canada dates back to 1948, in Saskatchewan. The federal government has conducted a national sediment program since 1961 in cooperation with the provinces, territories, and other interested agencies, such as hydroelectric companies. Data-collection techniques are standardized across the country to maintain data quality and comparability. Provincial governments also collect sediment data either as part of a regular sampling program or for specific



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studies. Consulting engineers and planners, as well as university researchers, also carry out sampling for site-specific projects.

These data have been used extensively to address reservoir sedimentation, environmental impact assessment, sediment-associated contaminant transport, and other concerns.

Sediment data and information are available from a variety of sources. A large amount of data is contained in a national computer data base operated by Environment Canada. The data base contains historical and current data for about 750 stations throughout the country, about 300 of which are currently monitored.

The types of data that are stored in the national computer data base are as follows:

- suspended-sediment concentrations
- suspended-sediment loads
- suspended-sediment particle size
- bed load
- bed-load particle size
- bed-material particle size
- sediment quality

The data are published annually on CD-Rom and are available from Water Survey of Canada, Meteorological Service of Canada, Environment Canada, Ottawa, Ontario K1A 0H3, Tel.: (613) 992-7121, Fax: (613) 992-4288.

FRESHWATER SERIES A-8

Note: A resource guide, entitled *Let's Not Take Water For Granted*, is available to help classroom teachers of grades 5–7 use the information from the Water Fact Sheets.

FLOOD PREVENTION

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INTRODUCTION

Although people cannot do much about floods, they can influence the nature and extent of flood losses. The latter can be achieved through flood management policies and programs. The first section of this contribution describes

some of the causes of flooding in order to provide insight into why some flood protection and response policies might succeed in one area but fail in another. Since natural flood mechanisms exist and socioeconomic conditions vary throughout the world, a mix of responses is necessary to effectively deal with the flood problem. The second section of this contribution describes the nature of some specific flood protection activities. In this manner, the need for effective flood management is demonstrated.

CAUSES OF FLOODING

Hydrometeorological Mechanisms

Flooding is caused primarily by hydrometeorological mechanisms, acting either individually or in combination (1,2). The regional variability in the intensity of these mechanisms reflects the diversity of climate across the region. The relative importance of the hydrometeorological flood mechanisms varies considerably throughout the year because of their close link to climate. In particular, severe flooding may result when several of these mechanisms occur coincidentally, as can happen in the spring.

Snowmelt Runoff Floods. The melting of a snowpack that has accumulated during the winter months is a common flood type. This occurs in watersheds of all sizes, often in combination with storm-rainfall runoff and/or ice jams. The amount of snowmelt runoff is controlled directly by the thickness, ripeness, and extent of the snowpack and by the rate of melting. The greater the amount of snow on the ground during a melt period, the more water is potentially available for snowmelt runoff. Ultimately, however, the rate of snowmelt is controlled primarily by radiant energy (3). Relatively cool weather causes slow melting of the snowpack and the gradual release of meltwater, while warm weather speeds melting and releases meltwater more rapidly, thereby increasing the possibility of flooding. Severe flooding from snowmelt can happen when there is a rapid shift from cold to warm temperatures in the spring following a winter of high snow accumulation. It can also be triggered during the winter months by a sudden and rapid thaw. Since the climatic factors influencing both the accumulation of snow and the rate of melt are regional, snowmelt flooding commonly occurs over large areas and affects numerous watersheds.

Storm-Rainfall Floods. Heavy or torrential rainfall associated with convective storms and midlatitude and tropical cyclones (including hurricanes) can cause flooding, even after a period of drought. This type of flood can develop rapidly, particularly when the ground is saturated and the rate of rainfall far exceeds the capacity of the ground to absorb the water, or when a significant portion of the ground surface is covered with impermeable materials, such as concrete or pavement in urban areas. The magnitude of storm-rainfall flooding depends on the intensity and duration of the rainfall, the areal extent of the storm, prestorm ground moisture conditions, and drainage basin characteristics (e.g., topography, overburden thickness, vegetative cover, drainage density).

Storm-rainfall floods can significantly affect watersheds of up to 100,000 km² in size throughout the world. Rivers with larger watersheds generally are unaffected because the zone of intense rainfall within a storm system commonly is concentrated within a smaller portion of the drainage basin. However, prolonged, heavy rainfall over a large area from a succession of storms may produce storm-rainfall flooding along a river draining a very large watershed.

Rain-on-Snow Floods. Rain-on-snow floods are a combination of snowmelt runoff and storm-rainfall floods. They occur when rainfall runoff is augmented by snowmelt, which increases the amount of water flowing into a stream system. The rate of snowmelt during a rainstorm is affected directly by air and rainfall temperatures, amounts of rainfall, and wind speeds. However, the amount of surface runoff released from a snowpack is controlled by snowpack ripeness, since an unripe snowpack will store more rainwater than a ripe snowpack. When heavy or sustained rainfall occurring in combination with other meteorological conditions leads to ripening and significant melting of a moderately deep snowpack, rain-on-snow can generate substantial runoff causing very severe flooding (3).

Ice Jam Floods. Ice jam floods result from the temporary obstruction of river flow by the buildup of ice fragments within the channel and can occur during both the freeze-up and breakup periods. However, ice jams during breakup are more likely to cause flooding. Once formed, an ice jam causes the river to rise immediately upstream and may overtop its banks, depending on the height of the obstruction relative to the sides of the channel. The failure of an ice jam can release a surge of water and ice downstream that causes a sudden rise in water levels and flow velocities downstream. Ice jams that form during the breakup period commonly coincide with the freshet flow arising from snowmelt runoff and can accentuate the level of flooding. Flooding from ice jams tends to be localized since it is dependent directly on the formation of an ice jam, and the tendency for an ice jam to form at any given location along a river is variable. Ice jam formation along a river is promoted by the presence of local sections of intact ice cover during breakup and/or local channel characteristics (e.g., channel shoaling, variation in channel width, channel splitting by islands or bars, and sharp bends). However, ice jams can also form behind bridges and other artificial structures that constrict a channel. The severity of ice jamming varies from year-to-year and depends on factors such as the harshness of the winter, the amount of ice decay and melting prior to breakup, and the amount of rise in river level immediately prior to and at the time of breakup. Ice jams are commonly associated with larger, north-flowing rivers.

Natural Dams

Floods can also be caused by the formation and failure of natural dams, although these events are far more localized and less frequent (at the national scale) than hydrometeorological flooding. Floods from natural dams occur due to the blockage of drainage by landslides,

glaciers, and moraines. Flooding occurs upstream of the natural dam as a result of ponding, but also downstream if there is a failure of the dam (or in some instances the development of a tunnel under a glacier) that allows the rapid drainage of the impounded water. These “outburst floods” produce peak discharges that are proportional to the volume of the impoundment, rather than the area of the contributing watershed, and the resulting flood can be larger by an order of magnitude or more than the maximum expected hydrological flood for the stream. Such large floods consequently may cause enormous erosion and channel change along the flood paths for many kilometers downstream of a dam and represent a much greater potential risk than the flooding behind the dam. With some specific glacier dams, outburst floods have happened nearly annually over periods of up to several decades because the dam has reformed repeatedly after successive drainages of the impoundment.

FLOOD MANAGEMENT

Preventing flood damages is one important element of flood management. There are a large number of strategies and methods available today to address flood hazards and disasters. Floods are natural environmental events (2). They present a hazard and may cause a disaster only after humans construct an environment on flood-prone lands (1,4–6). Building structures within floodplains is a development strategy chosen frequently by many societies all over the world. As a result, floods and floodplains need to be “managed” (7). Today, the common framework for dealing with floods comprises four general types of activities: (1) modifying flooding, (2) modifying susceptibility to flooding, (3) modifying the impacts of flooding, and (4) preserving the natural and beneficial functions of floodplains. Modifying flooding refers to structural flood control measures that are designed and constructed to keep the flood hazard away from people and buildings. Modifying susceptibility to flooding refers to activities designed to keep people and buildings away from the flood hazard. These are generally called nonstructural mitigation measures and include a broad set of tools (8,9). Modification of impacts of flooding on people includes floodplain management activities. These activities do not reduce the amount of damage that floods might cause, but they help “spread” the costs associated with those damages. The fourth set of management activities seeks to preserve the natural and beneficial functions of floodplains from an ecological perspective.

Flood Management Process

The flood management process can be divided into three major stages: planning, flood emergency management, and postflood recovery (10). During the *planning* stage, different alternative measures (structural and nonstructural) are analyzed and compared for possible implementation in order to reduce flood damages in the region. The analysis of alternative measures involves project formulation for each measure, understanding advantages and disadvantages of alternative project arrangements, evaluation

of positive and negative project impacts, and finally relative comparison of alternative measures. Flood *emergency management* includes regular appraisal of the current flood situation and daily operation of flood control works. A very important aspect of the appraisal process is identification of potential events that could affect the current flood situation (such as dike breaches, wind setup, and heavy rainfall). At this stage decisions are made on urgent major capital works and flood upgrading measures for existing structures. From the appraisal of the current situation information is initiated regarding evacuation and repopulation of different areas. *Postflood recovery* involves numerous hard decisions regarding return to the “normal life.” Issues of main concern during this stage of the flood management process include evaluation of damages, rehabilitation of damaged properties, and provision of flood assistance to flood victims. At this stage also, all environmental impacts are evaluated and mitigation strategies selected.

Structural Flood Management Measure

Flood management has the objective to evaluate utilization of possible methods for reducing flood damages to existing buildings and other facilities and reducing flood risk to permit additional growth. Structural measures can be further divided into local protection measures and upstream flood protection measures (usually storage reservoirs). Structural measures emphasize construction of levees or walls to prevent inundation from floods below some specific design flood flow. Additional works may include drainage and pumping facilities for areas that are sealed off from precipitation runoff to the river by the levees; diversion structures to divert flow during the peak from the protected region; channel modifications to increase the hydraulic capacity or stability of the river; and one or more reservoirs upstream from the protected area to capture the volume of a designed flood and release it at nondamaging rates.

Nonstructural Flood Management Measures

Nonstructural measures include zoning (to limit the types of land uses permitted to those that may not be severely damaged by floods), protection of individual properties (e.g., waterproofing of the lower floors of existing buildings), a flood warning system (to evacuate residents and to move valuables), and flood insurance (to recognize the risks of floods and to provide compensation when damages are not avoidable at acceptable cost) (11,12).

Responding to Floods and Sharing the Risks

As no flood risk management program can provide absolute protection, it is also appropriate to examine the institutional arrangements for responding to and recovering from floods. In most countries there are important federal flood management agencies. They usually coordinate and encourage emergency preparedness activities within the federal government, and between federal and lower level governments. In the context of flood response, as for all disaster response, they place initial responsibility upon individuals. Based on the extent of the flood and on an

individual's capacity to respond, responsibility can move from municipal, to regional, and finally to federal levels. Each level of government must request the support of the next one. Emergency preparedness and response is clearly a shared activity among individuals, the private sector, and all levels of government.

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EFFECTS OF DDT IN SURFACE WATER ON BIRD ABUNDANCE AND REPRODUCTION—A HISTORY

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Early in the DDT controversy so many allegations were made that it was difficult to respond to them all in the available time. Hundreds of scientists patiently refuted those allegations, however, and their contributions are summarized in many books and articles. The news media were notified that they had been misinformed by some environmentalists, and a few carried stories that agreed more with the facts. They also became aware that most scientists disputed the allegations that DDT caused cancer, mutations, hepatitis, poliomyelitis or other human illnesses. It was made clear that DDT residues did NOT “build up” in animal food chains, because they were metabolized or excreted by fish, birds and mammals.

BIRD POPULATIONS THREATENED?

Some persons attempted to link bird declines with the heavy usage of DDT between 1946 and 1960 and to imply that birds later increased after DDT was banned. When those claims were investigated with objectivity, it became evident that most declines had occurred either before DDT was present, OR after levels of DDT had waned. The Environmental Defense Fund commented on the abundance of birds during the DDT years, with “increasing numbers of pheasants, quail, doves, turkeys and other game species.” To the list of increasing bird populations, the Audubon Society added eagles, gulls, ravens, herons, egrets, swallows, robins, grackles, red-winged blackbirds, cowbirds, and starlings. The annual Audubon Christmas Bird Counts between 1941 (pre-DDT) and 1960 (after DDT use had waned) showed that at least 26 different kinds of birds became more numerous during those decades of greatest DDT usage. They documented an overall increase from 1,379 birds seen per observer in 1941 to 5,742 birds seen per observer in 1960. Statistical analyses of the Audubon data confirmed the perceived increases.

At Hawk Mountain, Pennsylvania, counts of migrating raptors were made daily by teams of ornithologists for more than 40 years. The results were published annually by the Hawk Mountain Sanctuary Association and revealed great increases (not decreases) of most kinds of hawks during the DDT years. Total numbers of hawks counted there increased from 9,291 in 1946 to 13,616 in 1956 and 20,196 by 1967. Sparrow hawks seen were as follows: 1946–98; 1956–192; 1965–408; and 1967–666.

The activists could not explain the population explosions of white-tailed kites in heavily-sprayed California, or of robins, starlings, cowbirds and red-winged blackbirds throughout North America. They tried, saying that not all birds were affected by DDT, but only predators. Because it was evident that predators were also multiplying rapidly, they decided that not all predators were affected, but only those that preyed on fish. But the fish-eating murrelets, ospreys, and gulls were rapidly increasing, too. The murrelets on Funk Island (along the North Atlantic Coast) increased during the DDT decades, from 15,000 pairs in 1945 to a million and a half pairs by 1971, and gannets there went from 200 pairs in 1942 to 3,000 pairs in 1970 (1).

Herring gull numbers in Massachusetts exploded during the DDT decades, from 2,000 pairs (1941) to 35,000 pairs (by 1971). Although they were on the “protected” list, Massachusetts gave the Audubon bird-lovers permission to poison 30,000 of them on Tern Island. The local Audubon president, William Drury, said “It’s kind of like weeding a garden” (2).

Many other kinds of birds multiplied so well in the presence of DDT and DDE that they became destructive pests. In North Carolina, 6 million blackbirds ruined the town of Scotland Neck in 1970, polluting streams, depositing 9 inches of droppings on the ground, and fouling the forests where they roosted at night. Near Fort Campbell, Kentucky, 77 million blackbirds were roosting, and the resulting threat of human disease (histoplasmosis) was of great concern (3). When marshes were sprayed in the Midwest to control mosquitoes, a common result was reported: “Today in a small area of northern Ohio, 10

million redwings mill about in the cornfields after the nesting season.” They destroyed the crops, while farmers begged for avicidal relief. In Virginia, the Department of Agriculture stated, “We can no longer tolerate the damage caused by redwings. . . 15 million tons of grain are destroyed annually—enough to feed 90 million people” (4).

Rachel Carson started environmental activists down the path to extremism when she wrote *Silent Spring* in 1962 (5). Later, the groups were joined by extremists opposed to technology and dedicated to reducing human populations in Third World Countries in every way possible, including insect-transmitted diseases.

In the front of her book, Rachel Carson wrote that it was “Dedicated to Dr. Albert Schweitzer, who said ‘Man has lost the capacity to foresee and to forestall. He will end by destroying the earth.’” Miss Carson certainly knew he was referring to atomic warfare when she quoted that, but she implied that he meant there were hazards from pesticides such as DDT. Because I had found so many other errors in her book, I got a copy of Schweitzer’s autobiography to see if he really mentioned DDT. He did, writing on page 253: “How much labor and waste of time these wicked insects do cause us. . . but a ray of hope, in the use of DDT, is now held out to us” (6).

She also wrote (5, p. 118): “Like the robin, another American bird [the bald eagle] seems to be on the verge of extinction.” Yet the robin was never endangered at all! Roger Tory Peterson, America’s leading ornithologist, also writing in 1962 (7), stated that the robin was “the most abundant bird in North America”. There is no doubt as to who was correct! The eagles had declined before DDT, but they increased during the DDT years. In addition, the Audubon Society’s Christmas Bird Counts revealed that twelve times more robins were counted, per person, in the 1960 count than were seen during the 1941 count (before DDT). In addition, there were 21 times more cowbirds, 38 times more blackbirds, 131 times more grackles, and so forth. No matter how deceitful her prose, the influence of Carson’s *Silent Spring* was great, and it continues to shape environmentalist propaganda as well as U.S. policy.

I taught medical entomology for years and was concerned about malaria, yellow fever, typhus, encephalitis, river blindness, leishmaniasis, and other tropical diseases that were transmitted by blood-sucking insects and could be controlled only by insecticides. The National Academy of Sciences wrote in its 1970 book, *The Life Sciences*, that “In little more than two decades DDT has prevented 500 million deaths due to malaria and other diseases, that would otherwise have been inevitable” (8). I found no references to those diseases in *Silent Spring*, however she wrote (5, p. 187): “Only yesterday mankind lived in fear of scourges of smallpox, cholera and plague. Now our major concern is no longer with the disease organisms that once were omnipresent; sanitation, better living conditions and new drugs have given us control over infectious disease. Today we are concerned with a different kind of hazard that lurks in our environment.” She then launched into many pages of unsupported attacks on insecticides, with never a mention of those millions of human lives that had already been saved by DDT!

Charles Wurster was the chief Environmental Defense Fund biologist. He got some tanks of sea water, put some marine algae in them, and poured large amounts of DDT into the tanks, to see if it would harm the algae. He described how ethanol was added to each flask of DDT, “in order to yield the desired concentrations [500 parts per billion].” The solubility of DDT in water is only 1.2 parts per billion, so we wondered about the validity of his research, exposing the algae to hundreds of times more DDT in solution than would be possible without the alcohol. In previous tests with the same kinds of algae, Ravenna Ukeles and others had found that 600 ppb of DDT in the water did not affect the algae.

Wurster published his non-scientific results in *Science* magazine (9). In 1969 Dr. Paul Ehrlich expanded on Wurster’s hoax and wrote what he called a “scenario” based upon it. He titled his article “*Ecocatastrophe*,” and published it in *Ramparts* magazine (10). His article began with:

“The end of the ocean came late in the summer of 1979, and it came even more rapidly than biologists had expected. There had been signs for more than a decade, commencing with the discovery in 1968 that DDT slows down photosynthesis in marine plant life. It was announced in a short paper in *Science*, but to ecologists it smacked of doomsday. They knew that DDT and similar chlorinated hydrocarbons had polluted the entire surface of the earth, including the sea.”

Thousands of school children were required to read Ehrlich’s scenario. It was a surprise to see that same exact article published by Ehrlich in England a year later, in a journal titled “*The Year’s Best Science Fiction*.”

BIRD DATA

Although at first many anti-DDT activists said that DDT was killing off adult birds, they soon had to admit it was not. They were not “dropping from the sky dead,” as alleged by some environmentalists (5, p. 124), (11), nor were they “falling out of the trees by the thousands,” as other extremists wrote (12).

Wurster (13) referred to “increasing numbers of pheasants, quail, turkeys, and other game species,” and Audubon naturalists added at least 26 other species that increased in numbers during the decades of greatest DDT use (14,15).

The propagandists changed their story, and said DDT was killing nestling birds. That was shown by tests to be untrue, when all of the food given to experimental nestlings contained DDT, but no adverse effects resulted (16). They next said perhaps DDT makes the eggs thin-shelled or kills the developing embryos. Legitimate tests proved that they were wrong again.

In their effort to make people believe that DDT in normal bird diets caused thinning of eggshells, the environmental industry fostered tendentious feeding experiments during which their birds were exposed to treatments that every poultryman knew would cause severe eggshell thinning. This included removing calcium from the food, starving the birds, depriving them of water, and cutting the light from 16 hours a day to 8 hours a day.

Obviously, heavy DDT usage was not eradicating birds of any kind but what was causing the great *increase* of birds in areas that had been sprayed with DDT? Causes included the reduction in numbers of blood-sucking insect parasites, which provided birds protection from avian malaria, rickettsial-pox, bronchitis, Newcastle disease and encephalitis. Also, more bird-feed seeds and fruits were available for birds after phytophagous insects had been killed by insecticide sprays. Perhaps even more important was the medicinal effect of DDT. When ingested, DDT triggered the induction of hepatic enzymes that detoxified potent carcinogens such as aflatoxins produced by *Aspergillus* fungi, which abound in the natural foods eaten by birds. Aflatoxins are carcinogenic at levels of 0.03 to 0.08 ppm in the diets of birds, fish and mammals. Many humans in third world countries also die because of aflatoxins in their food.

The anti-DDT extremists relied on just three or four studies in their attempt to convince the Environmental Protection Agency hearing examiner that DDT causes serious thinning of the shells of birds. The studies included Bitman et al. (17), and Porter and Wiemeyer (18). Of course they could not explain the thin shells that occurred before DDT was used, nor could they explain the lack of adverse effects on eggs of captive birds that were experimentally dosed with DDT.

Research by Bitman was designed in such away that thin eggshells would have resulted regardless of whether DDT was added to the diet. His Japanese quail were fed a diet containing only 0.56% calcium “to provide calcium stress during egg-laying.” The DDT concentration added to the diet was 4,000 times the concentration in normal human diets in the United States. They blamed the DDT, instead of the calcium deficiency, for the production of thinner shells.

Compare that with the 1970 tests by FWS researchers Tucker and Haegele (19). They fed different amounts of calcium to each of three different cages of quails, *without* DDT: 3% calcium to the “controls”, 1.73% calcium to another group, and 1% calcium to the third group. None got any DDT or DDE. The shells formed by the 1% calcium birds were 9.3% thinner than the controls. Remember, Bitman and his colleagues at the FWS laboratory fed their quail food with only 0.56% calcium! That would be expected to result in much thinner eggshells than did Tucker’s 1% calcium group, even if Bitman had not added any DDT or DDE to the food.

During Congressional testimony, I criticized Bitman’s calcium-deficient quail diets. He evidently understood my criticisms and subsequently performed additional research with his quail. This time he fed the quail 2.7% calcium in the diet, instead of 0.56%. The result was that both the DDT-fed and the DDE-fed birds produced heavier eggshells, and there was no difference in shell thickness between the DDT-fed birds and the ‘controls.’ He sought to publish those results in *Science* magazine, but this time they were rejected. He had to publish them, instead, in *Poultry Science* (20).

Why did *Science* refuse Bitman’s second article? The editor of *Science* had informed Dr. Thomas Jukes earlier that *Science* would never publish any article about DDT

that was not antagonistic to that insecticide, and through the years he enforced that attitude. Anti-DDT writers simply kept expanding their anti-DDT propaganda in *Science* magazine. It was written by the same coterie, and for references in *Science*, they simply kept citing each other. No other studies were accepted by America’s leading science journal. Without that sheltered bias, the case against DDT would have quickly folded and millions more human lives would have been saved.

BENEFICIAL RESULTS OF DDT

There was no scientific basis for the allegations that DDT caused bird populations to experience declines. On the contrary, it is obvious that proper use of DDT actually increased the populations of fish, birds and mammals, because of the following actions: (1) DDT applications increased the yield of plant products in fields, woods, marshes, fencerows, and roadside ditches; (2) they increased protective vegetative cover for birds and mammals and provided better nesting and denning sites; (3) they killed mosquitoes, flies, fleas, lice, and mites in sprayed areas, not only reducing stress and annoyance, but also preventing bird diseases transmitted by those arthropods (avian malaria, bronchitis, rickettsial-pox, Newcastle disease, and so forth); and (4) DDT stimulated the liver to produce more hepatic microsomal enzymes that broke down toxic substances in the body, including “aflatoxins” that occur commonly in natural foods such as moldy grain, nuts, cotton seed. Those aflatoxins, resulting from *Aspergillus* fungi, are among the most potent carcinogens known in foods. DDT-induced hepatic enzymes prevent the development of tumors in birds and mammals, according to research by McLean and McLean (21), Edward Laws (22), and Salinskas and Okey (23).

The anti-DDT propaganda was refuted by scientific research and also by the multiplying hoards of birds counted during the Christmas Bird Counts and the Hawk Mountain Sanctuary migration counts.

The anti-DDTers were apparently not impressed, even when birds became so abundant that they destroyed fields of corn or broke down the tree branches upon which they roosted. They didn’t seem to realize that DDT had failed to harm the destructive birds. They apparently felt that DDT caused good birds to decline and bad birds to have population explosions!

PERSISTENCE OF DDT RESIDUES

During the seven months of EPA hearings on DDT, Dr. George Woodwell testified regarding an article written by him and Charles Wurster (24). The hearing transcript contained a statement that: “DDT residues in an extensive salt marsh on the south shore of Long Island average more than 13 pounds per acre” (This was discussed on p. 7232 of the hearing transcript, on January 13, 1972), as follows:

The USDA attorney asked Woodwell:

Q “Isn’t it a fact that after you initially studied this marsh you continued your samplings, and found as a

result that you were getting an average of one pound per acre of DDT?"

A "No, I wouldn't agree with that."

Q "Didn't you also find out later that one of the areas where you took your samples was an area that was convenient for the mosquito Commission's spray truck, too?"

A "I can't say that I discovered that, either. I don't believe there's any evidence to that effect."

Q "Dr. Wurster, perhaps?" [We had told the attorneys the details of Wurster's Seattle testimony, and Wurster was in the audience, this day.]

A "I don't believe he knows that, either. I don't believe there's any evidence to that effect."

Q "Dr. Wurster, your coauthor, made the following statement at the Washington state hearings, and I'm quoting him verbatim: He testified: 'We have since sampled that marsh more extensively, and we found that the average in the marsh was closer to one pound per acre. The discrepancy was because our initial sampling was in a convenient place, and this turned out to be a convenient place for the mosquito Commission's spray truck, too.' Did you learn that after the fact, Doctor?"

A "That is a true statement in my experience. I did not know that Dr. Wurster had said that, but that is a true statement."

Q "Doctor, have you ever published a retraction of this 13 pounds per acre, or a further article which discloses the results of your further sampling which brings the average down to around one pound per acre?"

A "I never felt that this was necessary."

(Woodwell also admitted they had only taken six samples of the soil in that extensive marsh!)

BREAKDOWN AND DISAPPEARANCE OF DDT

Earlier, George Woodwell et al. wrote (25), that "6 billion pounds of DDT had been used, but only 12 million pounds could be accounted for in all of the earth's biota", and that was "less than a thirtieth of one year's production of DDT in the 1960's." He wrote that "most of the DDT has either been degraded to innocuousness or sequestered in places where it is not freely available." That contrasted so sharply with his testimony during the EPA hearings that a reporter asked him why he had completely omitted it from his testimony. Woodwell replied that the EPA lawyers told him not to mention the article, "lest my testimony be disallowed" (26).

A Department of Interior study at Gulf Breeze, Florida was reported in *Chemical Assays* (27). Huge glass containers were filled with sea water, a large amount of DDT was added, and the closed containers were suspended in the Gulf. Nearly every day, a sample was taken out for analysis. After 38 days, 92% of the DDT (and its metabolites, DDD and DDE), had disappeared.

In Washington state estuaries, the Bureau of Commercial Fisheries monitored pesticide residues in shellfish at

19 stations during 3 years of heavy DDT use (1966–1969). Ninety-three percent of the samples contained less than 10 parts per billion of DDT (and the highest level found was only 0.1 part per million). Only 3% of the samples contained chlorinated hydrocarbon insecticides. Shellfish are known to "concentrate" chlorinated hydrocarbons in their systems at levels 40,000 to 70,000 times as great as that in the surrounding water, so it is evident that the coastal waters were very free of DDT residues.

DDT is broken down rather rapidly by heat, cold, moisture, sunlight, alkalinity, salinity, soil micro-organisms, and a great many other things. Studies abound regarding DDT disappearance from soil, sand, leaf litter, plant tissues, animal tissues, and water.

Crocker and Wilson applied DDT to a tidal marsh (28). In less than 24 hours, only traces remained, and even those traces disappeared in five days.

Finally, the environmental extremists had to resort simply to endlessly repeating all of the old allegations and hoping that most people had not heard about the refutations. The failure of the news media to report the truth helped them get away with that ruse, and pseudo-environmental magazines continued to frighten readers with the same old unsubstantiated myths.

BROWN PELICANS

In 1918, T. Gilbert Pearson and Robert Allen investigated habitats of brown pelicans along the 1,500-mile coastline of the Gulf of Mexico from Key West to the mouth of the Rio Grande. Belatedly reporting his results in *National Geographic* magazine in March 1934, (29), Pearson estimated a total of 65,000 pelicans during that survey. Robert Allen repeated that Gulf survey in 1934, after becoming president of the National Audubon Society. He discovered there had been an 82% decrease in pelican numbers in the 16 years since their earlier survey. He saw only 900 pelicans in Texas and practically none in Louisiana (30). In 1937, the results were published in *Adventures in Bird Protection* (31). According to *Conservation In The United States* (32, p. 414), 17 colonies existed along the Texas coast in 1918, but only one colony of 500 pelicans remained in 1939, and it declined to 200 by 1941.

I reported on these and dozens of other findings during my testimony in Congressman W.R. Poage's hearings for the House Agriculture Committee, on March 18 1971. Finley and Keith (of the U. S. Fish & Wildlife Service), however, were issuing bulletins stating that "A catastrophic decline has occurred on the Gulf Coasts of Texas and Louisiana where a population of over 50,000 brown pelicans has all but disappeared since 1961." The cause, they insisted, was DDT!

In 1961, van Tets saw the last active nesting colony in Louisiana, on North Island, containing only 200 pairs (33). Those, plus the 900 reported in Texas, fell far short of the alleged 50,000 conjured up by Finley, so I called him to ask for details. He answered on March 29, 1971, saying "In response to your inquiry, I have investigated the source of the reported 50,000 brown pelicans on the Texas and Louisiana coasts. The only comprehensive estimate was

made by T.G. Pearson: 65,000 birds, in his 1934 book.” Was it possible that Finley didn’t know that the total in 1918 was for the entire 1,500-mile Gulf Coast and was in *National Geographic*, instead of a book? Was it also possible that he was unaware of Robert Allen’s 1934 survey? He cited only five references, dated 1879 to 1955, and concluded “Although the reports are sketchy, Jim Keith and I both feel that the estimate of 50,000 is not unreasonably high.” Five months later, however, (August 2, 1971), he wrote to Congressman Poage, stating: “The year 1961 was merely a hasty approximation of an unknown time. After reviewing the evidence, I think now that I should have said the 50,000 pelicans disappeared **by** 1961 [instead of **since** 1961].” (emphasis added). The truth was that the total number did not exceed 300 pelicans along the entire Texas coast in 1942 (before DDT), according to Audubon researchers. If the pelicans had somehow multiplied from less than 500 in 1959 to 50,000 before 1961, as Finley and Keith claimed, they would have had to do it in a habitat that was rich in DDT!

The significance of Finley’s belated retraction of the untruthful “50,000 pelican disappearance” was great. It removed the possibility that DDT could have caused a pelican disappearance that never occurred! In fact, the declines obviously occurred long before any DDT was present, and resulted from human persecution, oil spills, gas blowouts, bombing, machine gun practice by the military, and hurricanes.

In 1968, Schreiber and DeLong surveyed California’s Anacapa Island pelican colony for the Smithsonian Institution and wrote: “While fluctuations in numbers do occur, there has been no apparent decline in recent years along the California coast” (34). (There were no adverse effects from DDT).

In 1968, Jehl and Keith saw dozens of sick pelicans along the shore of the Gulf of California (35). Soon thereafter, coastal southern California experienced a severe epidemic of Newcastle Disease. A major symptom of that disease is the production of misshapen, soft-shelled eggs.

On January 28, 1969, the Santa Barbara Oil Spill occurred near the southern California coast. Oil surrounded Anacapa Island, and two biologists (Jehl and Risebrough) went out in March to see how much damage the oil was causing to sea lions. They also entered a brown pelican colony and it was reported in *Cry California* (36) that there were many soft or broken eggshells in and near the nests there. Jehl recalled that “As soon as the [poor] condition of the eggs was appreciated, Risebrough was able to postulate that chlorinated hydrocarbon pesticides [DDT] were probably responsible” (37, p. 18).

In California, the pelicans had experienced no difficulties during the 20 years of heavy DDT usage but suddenly suffered a reproductive failure just after the great oil spill surrounded their Anacapa Island colony, according to Jehl and Kennedy. Scientists should have been impressed by that, but the oil spill was not discussed in any reports about the pelican nesting failure.

The oil film, through which the pelicans were diving for fish, contained 21 ppm of mercury (38), and the anchovies upon which the pelicans fed contained 17 ppm of mercury.

Mercury is known to have in serious adverse effects on birds, their eggs, and their young.

Environmentalists concealed the fact that California Fish & Game biologists had found that those anchovies contained high levels of lead, which is known to cause severe shell-thinning. They collected hundreds of pelican eggs from that colony during the next two summers, and their shells were measured with screw micrometers. In April 1972, I obtained all of those measurements and found that they clearly revealed inverse correlations between DDT residues and shell thicknesses. Some of the thinnest shells were those of eggs with low DDT levels, and the higher DDT concentrations were often in the thicker-shelled eggs (39). That was also reported by Switzer at the 1972 EPA hearings on transcript pages 1812–1836. I had discussed all of this during my testimony before Congressman Poage, and it was published as a document by the *Committee on Agriculture*, March 18, 1971 (40). Robert Finley, who was in charge of Fish & Wildlife Service pelican studies throughout the United States, responded to my testimony before Congressman Poage’s Committee by writing Poage (August 2, 1971) that “there is not a shred of evidence that spilled oil is capable of causing thin-shelled eggs or otherwise affecting bird reproduction.” On the contrary, Hartung (41) said that spilled oil reduces hatchability by 68%. Anonymous reports in *Conservation News* (NWF) (42) discussed “Embryo mortality from oil on feathers of adult birds.” Dieter wrote that “oil on eggs kills 76 to 98% of embryos inside, and birds that have ingested oil produce 70% to 100% fewer eggs than normal” and “Liver and spleen atrophy may result from oil exposure.” (43). King (44) said, “oil is a cause of pelican mortality for six weeks after a spill.” Szaro (45) reported that “oil causes kidney nephrosis, pancreatic degeneration and lipid pneumonia.” Birds may lose their ability to compensate for osmotic water loss when oil inhibits water uptake by the intestine, which in turn fails to trigger the necessary desalination of water by the nasal glands. He concluded that “It is possible that dehydration may cause the death of many oil-contaminated birds.”

In 1923, Dawson described, in *Birds of California* (46), how “merely the brief presence of a human on Anacapa caused adult pelicans to spring from their nests, breaking their eggs with their feet as they did so, and the exposed eggs and nestlings were quickly eaten by marauding gulls.” All such warnings were ignored by the Fish & Wildlife specialists! The FWS biologist in charge (Jehl) even admitted that **they had shot-gunned pelicans sitting on their nests of eggs** (47), so they could compare the DDT residues in the birds with those in their eggs. They also admitted that they had **collected 72% of all intact eggs in the Anacapa pelican colonies for analysis!** Those analyses revealed an inverse statistical correlation between DDT residues and shell thickness! When the government biologists again invaded the Anacapa colony in 1969, the results must certainly have been anticipated, yet they sought to blame the nesting difficulties entirely on “traces of DDT in the fish” eaten by the birds.

My student and collaborator, Richard Main, was employed by the National Park Service for 2 years. He trapped many large Norway rats beside pelican nests in the Anacapa colony (in "safe traps"). The National Park Service wanted to poison the rats, to protect the young pelicans, but that was prohibited by the California Fish & Game Department because, they said, they feared that native rodents there might also be killed. Several feral cats also roamed about in the colony because a man named Frenchy left them there when he moved from Frenchy's Cove a few years earlier.

Franklin Gress, a graduate student at the University of California, studied the Anacapa pelicans in 1970. The desertion of the colony occurred following his flights to the island by helicopter and his hours sitting in nests in the colonies. His extensive notes were released in 1970 by the Resources Agency of the California Department of Fish & Game. His first visit was March 29, when he counted 127 nests, but he wrote that "The number of nesting birds dwindled, as nests were deserted, until by April 1st the colony was virtually abandoned." He returned on April 20th, and soon wrote that "the nesting attempts seemed to be irresolute." Two days later "the colony was suddenly deserted." A similar result followed his third visit, on June 1st, when he noted that "It was evident that the process of abandonment had begun." By June 20, "the colony was nearly abandoned, and there was no other nesting on the island." On June 29, he returned and reported that "only one nest contained a nestling... this was the only pelican to hatch in either colony on Anacapa Island in 1970." (It should be noted that the Park Rangers who patrolled the island by boat laughed and said they saw numerous young pelicans standing on nests during the intervals between Gress' disruptive visits.)

In 1971, at the request of the National Park Service and Secretary of the Interior William Pecora, Richard Main and I drew up protective regulations for the Anacapa pelican colonies, keeping the destructive government biologists out of the colony during the following breeding season, from March 15 to July 31. The oil was mostly gone, and there had not been time enough for DDT levels to decrease in the water or in the fish, yet the pelican productivity was great! The pelicans quickly recovered their breeding potential (48,49).

The Federal Fish & Wildlife Service and the California Fish & Game Department continued to issue pelican reports that never mentioned the oil spill, the rats, the disruptions by humans and helicopters, the egg predation by biologists, the lead in the pelican bones, or the epidemic of Newcastle Disease in coastal Southern California. They always attributed the alleged declines, however, to DDT!

Those allegations played a very important part in the EPA hearings that led to the banning of DDT in December 1972. We, and many National Park personnel, were convinced that they wanted to totally destroy the colony which would help the EPA to ban DDT. (Malaria would then continue to destroy millions of humans in the Third World and halt the population explosions there.) As it turned out, they succeeded in the broader goal, but failed to actually destroy the pelican colonies.

The Fish and Wildlife Service never publicly corrected its erroneous "since 1961" propaganda, so the news media continued to say that DDT nearly caused the marvelous pelicans to become extinct during the 1960s. Perhaps many people still think that the pelicans were saved from extinction "because attorney William Ruckelshaus banned DDT in 1972!"

That was enough to shake our confidence in the Fish & Wildlife Service, the Environmental Protection Agency, their pelican propaganda, and their integrity!

THE ENDANGERED OSPREY

Long before DDT was used in this country, Hickey (50) attributed a 70% decline of eastern ospreys to "pole trapping" around fish hatcheries. In 1971, Chesapeake Bay ospreys had no problem with DDT or DDE, but "lipids of osprey eggs had PCB levels from 545 to 2270 parts per million" (51). In 1972 Hickey avoided mentioning that ospreys were abundant throughout the west, despite massive applications of DDT there (52). Studies in Wisconsin and Michigan revealed an increase in nesting osprey productivity, from 11 young in 1965 to 74 young in 1970, before the DDT ban. Because data from other regions were sparse or unavailable, many incorrectly assumed that the population crashes were uniform throughout North America, but actually Ornithologist Charles Henny said, "there may be more of the birds in the West than ever before. Their future appears good" (53). Henny estimated that there were 5,000 or more breeding pairs in North America, plus non-breeding adults and juveniles. He wrote that "The population declines of the 1960s, largely attributed to DDT, were not as widespread as originally thought."

After DDT was developed, Hickey ignored the pole trapping and shooting of ospreys and blamed only DDT for all bird declines. I was a member of the Hawk Mountain Sanctuary Association and was impressed by its published data on ospreys. At Hawk Mountain, Pennsylvania, the survey of migrating ospreys during the DDT years revealed the following results: 191 ospreys in 1946, 254 in 1951, 352 in 1961, 457 in 1967, 529 in 1969, and 630 in 1972 (54). It was difficult to believe the charges that DDT threatened osprey extinction, when more were migrating every year after the introduction of DDT, setting new all-time records in several of those years.

There were many reports of high levels of mercury in the fish upon which ospreys depended for food. (Mercury causes drastic thinning of shells of eggs.) It was alleged, however, that DDT and DDE had contaminated the fish world-wide, and that, as a result, fish-eating birds could not reproduce successfully! Chesapeake Bay ospreys had no trouble with DDT or DDE, but their eggs contained high PCB levels (51). That was shown to cause drastic thinning of eggshells, as well as many other adverse effects, yet environmentalists continued to place the blame solely on DDT, despite the fact that feeding birds high levels of DDT alone did not cause birds to produce such thin eggshells.

After DDT was banned, the Hawk Mountain Sanctuary Association *Newsletter* reported in April 1976 that "For

reasons we do not understand at all, the number of osprey going south is returning to something like normal, just 318 in 1974 and 279 in 1975.” (Notice how many fewer there were in the years after DDT disappeared!)

BALD EAGLES

These birds have suffered much at the hand of humans, but not because of DDT. In 1917, Alaska enacted legislation providing a bounty of 50¢ per eagle and increased the bounty, later. From that time until 1952, the state paid bounties on 128,000 dead eagles (more than \$100,000). An article in a 1921 issue of *Ecology* magazine (55) was titled “Threatened Extinction of the Bald Eagle.”

In the 1930s, there were no records of bald eagles in most of New England, not more than 10 in Pennsylvania, and perhaps 15 or 20 along Chesapeake Bay. *Bird Lore* magazine (56) stated, “this will give some idea of the rarity of the eagle in eastern United States.” That was 14 years before DDT was present, and the eagles surely could not have disappeared in anticipation of DDT!

In 1937, Bent wrote that “the bald eagle probably nested at one time over much of New England, but there are no recent authentic records of them nesting in the three southern states there.” (57).

Each year, thousands of bird watchers participate in the annual Christmas Bird Count for the National Audubon Society. In 1941 (before any DDT was used), they saw 197 bald eagles (14), but in 1960 (after 15 years of heavy DDT use), the count was 891 bald eagles seen (15). If those results had been the reverse, I might have also thought that DDT was killing them!

The Hawk Mountain Sanctuary censuses, tabulated almost daily throughout the years, showed that the number of bald eagles migrating over Pennsylvania more than doubled during the first 6 years of heavy DDT use (1946–1952). (Their summaries also revealed great increases in most other kinds of raptors during the “DDT years.”)

In 1943, in *Science News Letter* (58), it was stated “When the timber was cleared it was inevitable that the eagles had to go. The cities grew and fouled the rivers with sewage and industrial waste. The once-teeming fish population vanished. With their main source of food thus taken away, it was only natural that the eagles might vanish also.” Notice that was written years before DDT appeared in North America!

In America’s Eagle Heritage (59), the study of bald eagles in the Everglades is discussed. “In 1959 there were 24 active nests, which produced 18 young; in 1964 there were 51 nests, which produced 41 eaglets.” Notice the sizeable increase! The chief Everglades biologist reported that “The population is stable and is reproducing at a rate more than adequate to maintain its numbers (and) I know of no evidence that the region ever supported a larger number of nesting eagles.”

At the Patuxent Wildlife Research Center in Maryland (U.S. Department of Interior), autopsies were performed on all bald eagles found dead in eastern U.S. between 1960 and 1965. Coon et al. (60) stated that none of those deaths was blamed on DDT. Forty-six had been shot or trapped,

and 7 died of injuries from flying into buildings or towers. Between 1965 and 1980, they performed autopsies on 652 more bald eagles from across the nation. The major cause of death was always shooting and trapping, but there were also many deaths from electrocution or impact with solid structures. The scientists concluded that “the role of pesticides has been greatly exaggerated.” In 1972, the carcasses contained twice as much PCB as DDT, DDD, and DDE combined, and the brains had four times as much PCB” (61).

In 1983, New York State contained only three active eagle nests. Later, 150 bald eagles were imported from Alaska. Peter Nye (62) wrote, “My colleagues and I identified 80 locations in New York believed to have been bald eagle nesting sites between 1880 and 1960. By 1940 only a handful of pairs could be identified; yet [he said] the oft-mentioned culprit, DDT, wasn’t put into use until the last few nesting eagles were already struggling for their survival” Nye and his colleagues imported eaglets from Alaska beginning in 1976, including 49 in the first 3 years. Now a healthy population is there.

Postupalski (63) found no significant correlation between DDT or DDE residues and shell thickness in a large series of Bald Eagle eggs. However, Stickel reported that bald eagles produced eggs whose shells were 14% to 21% thinner after lead, mercury, or a small dose of parathion was added to their food (64).

Krantz et al. (65) analyzed bald eagle eggs from three states and found that eggshells from Florida were thinnest (0.50 mm), followed by those from Maine (0.53) and Wisconsin (0.55). Analyses of their DDT residues revealed an inverse correlation: the eggs from Maine contained 21.76 ppm, twice as much as those from Florida and five times as much as those from Wisconsin. Obviously, DDT and DDE were not correlated with that shell thinning.

In Wisconsin and Michigan, an increase in nesting bald eagle productivity was noted, from 51 young produced in 1964 to 107 in 1970. In 1966, Fish & Wildlife biologists fed large amounts of DDT to captive bald eagles for 112 days, then concluded that “DDT residues encountered by eagles in the environment would not adversely affect eagles or their eggs” (64). Other wildlife authorities attributed bald eagle reductions to “widespread loss of suitable habitat,” but said “illegal shooting continues to be the leading cause of direct mortality in both adult and immature bald eagles.”

To summarize, the bald eagle in the lower 48 states was on the verge of extinction in the 1920s and 1930s, long before DDT was discovered. They were shot on sight for fun, bounty, or feathers, trapped accidentally, killed by impact with buildings and towers, or electrocuted by power lines. Many more may have died because of the high levels of PCBs in their lipids and brains. Thousands have now been reared in captivity and released into the wild. The increased numbers of bald eagles are admired and protected.

The most surprising thing is that the environmental industry and the news media continued erroneously to attribute the increase in eagle numbers from 1946 to 1970 to just one thing. . . the 1972 ban on DDT!

EGGSHELL THINNING

Shell-less eggs or eggs with thin or distorted shells are the result of any of a great number of factors. Older birds produce thin-shelled eggs, as do some young birds, and perfectly normal eggshells become 5% to 10% thinner as the developing embryo withdraws calcium for bone formation. Even the same bird produces varying eggshell thicknesses as time passes. Higher temperatures and dehydration cause thinner shells to be formed, as does higher relative humidity. A reduction in illumination causes drastic changes in shell formation, and the duration of the light has physiological effects as well as regulating the amount of food the birds have time to eat. Stress from excitement, fear, and noise is very destructive to egg-laying success. Simple restraint interferes with the transport of calcium throughout the bird's body, preventing it from reaching the shell gland and forming good eggshells. Lower than normal temperatures reduce the carbonic anhydrase activity, which causes thinner shells.

A great many environmental substances inhibit a bird's ability to produce normal eggshells (oil, lead, mercury, PCBs, 2,4-D, cadmium, lithium, chromium, selenium, and sulfur compounds in the diet). Sulfa drugs, especially sulfanilamide, may result in the complete absence of a shell around the yolk.

Tucker (66) reported that after water was withheld from quail for 36 hours, they produced eggshells 29.6% thinner when drinking was resumed. He also found that parathion ingestion caused 4.8% thinner shells, Sevin caused 8.7% thinning, mercury caused 8.6% thinning, and lead caused 14.5% thinner shells, but *p,p'*-DDT and DDE caused NO shell thinning and *o,p'*-DDT caused shells 0.5% thicker to be formed.

Many bird diseases, such as avian bronchitis, avian malaria, rickettsia pox, or Newcastle disease, may cause great thinning of eggshells. Such information has been well covered in the literature, including a great book, *The Avian Egg*, by the Romanoffs (67). The environmental extremists, who sought to blame DDT, never mentioned that book or Romanoff's *The Avian Embryo* (68).

A deficiency of phosphorus or Vitamin D also results in thin eggshells, but the most notorious cause of thinning is the lack of sufficient calcium. When the diet contains less than 1% calcium, the bird's leg bones may soften to such an extent that even walking is difficult. Wasn't it curious that the FWS researchers at Patuxent deliberately fed their DDT-dosed birds only food that was deficient in calcium, and then attributed the shell thinning to the massive amounts of DDT and DDE they added to the diets?

Dozens of researchers (and thousands of farmers) found that DDT did *not* cause shell-thinning. Jefferies fed his finches 50 to 300 micrograms per day of DDT, which "caused calcium metabolism to tend toward the production of *heavier* shells." (emphasis added) After 2 years they still formed shells that were 7% thicker (69). Frank Chermis fed quail 200 mg/kg of DDT for months, with no effect on shell thickness through four generations. DeWitt (70) fed his pheasants diets containing 50 ppm DDT throughout the year, and reported that the birds produced 23% more hatchable eggs than the controls (and 2 weeks later 100%

of the DDT chicks were alive, but 6% of the "controls" had died).

It was demonstrated repeatedly in caged experiments that DDT, DDD, and DDE did not cause shell-thinning, even at levels hundreds of times greater than wild birds would ever accumulate. During many years of carefully controlled feeding experiments, Scott et al. (71) found "no tremors, no mortality, no thinning of egg shells and no interference with reproduction caused by levels of DDT which were as high as those reported to be present in most of the wild birds where 'catastrophic' decreases in shell quality and reproduction have been claimed." Scott et al. also found that, "DDT did not have any deleterious effect upon the sex hormones involved in egg production and indeed may have had a beneficial effect upon egg shell quality."

Early egg collectors routinely saved only thick-shelled eggs. Comparisons between the shell thickness of museum eggs and those collected more recently are misleading and have led to much of the eggshell propaganda during the DDT years. Hickey and Anderson (72) measured the shells of 1729 museum eggs, and reported differences in shell weights of the eggs of five bird species, but said "changes did not occur in the five other species we studied, or in dozens of others, not studied." Four of the five species they reported in their big table had eggs that were *heavier* after DDT use began than those produced during pre-DDT years. Golden eagle eggshells during the "DDT years" were 5% thicker than those produced before any DDT was used. Red-tailed hawk eggs just before DDT was present had much thinner shells than did the pre-1937 eggs, but during the years of heavy DDT use, the birds produced shells that were 6% thicker.

Claus and Bolander in *Ecological Sanity* (73) wrote,

"We have nowhere seen reports of changes in the shell thickness of any species which exceed 50 to 60 microns (0.05 to 0.06 mm) and in some cases differences of 5 to 7 microns have been reported as 'significant decreases' (0.005 to 0.007 mm differences). Since only the 2nd and in some cases the 3rd decimal point differ, one wonders how significant those differences really were!"

Morris et al. (74) reported that "the most critical area of light intensity change is in the range between 8 and 16 hours and decreasing photoperiods depress rate of lay." It was interesting that Peakall (75) deliberately used those precise hours in his efforts to incriminate DDT. Peakall kept his ring-doves in a cage on a 16-hour light and 8-hour dark schedule. When ready for the experiment, he put the female of each pair into isolation and abruptly reduced the light to only 8 hours per day. At the same time, he began feeding the doves 10 ppm of DDT in all of their food! Poultry would have stopped laying entirely under those conditions, but the doves continued to produce good eggs. Apparently determined to stop their productivity, Peakall then began to inject intraperitoneally large amounts (150 mg/kg) of DDE (not DDT!) into each bird! (They still continued to lay eggs.)

Hauser (76) warned that "Providing adequate light is just as important as vaccination, parasite control, and feeding programs. When lightbulbs in the hen-house are

dirty and the hen receives less light than needed, the stress may be severe enough to cause a premature molt.” (The effect of Peakall’s 50% reduction in illumination would obviously be severe!)

Details of such “experiments” were exposed and discussed in detail by many qualified biologists, however, the media, the pseudoenvironmentalists, and the financially-rewarded anti-DDT “researchers” failed to publicize frauds. Consequently, the general public remained misinformed about the causes of eggshell thinning, and anti-DDT extremists continued to accumulate money and power as a result.

Measuring Eggshell Thickness

The great book by Claus and Bolander (73) includes nearly 300 pages dealing with DDT frauds. They revealed many examples of the untruthfulness of anti-DDT propagandists. A section titled “The Turning of the Screw” (73, pp. 400–405) exposes the reasons that screw micrometers failed to measure bird eggshells accurately. The differences between thicknesses of shells were measured “to the nearest 0.01 mm” (10 microns) and were, in some cases, reported as “significant” thinning if the difference was 5 to 7 microns! Claus and Bolander reminded readers that a human red blood cell is about 8 microns across and a human hair may be 50 to 80 microns in diameter. Also, an eggshell that has a soft texture and an easily-compressible inner membrane might not be measured very accurately with a screw micrometer by a person seeking to prove that DDT caused eggshell thinning.

POLYCHLORINATED BIPHENYLS

There are known to be about 200 different PCBs in the environment. They came into usage in the 1930s as plasticizers on wires and cables and for protective coatings on metals, plastics, and wood. They are also paint extenders and heat exchange fluids in transformers. Every fluorescent light ballast contained liquid PCB, so gas chromatograph analyses for DDT performed in rooms with those lights distorted all results. The plastic tubing in the apparatus, and other plastics that had been in contact with the samples also resulted in false analyses of DDT. Samples to be analyzed were frequently stored in plastic or cellophane containers, which also produced PCBs. In 1985, Wolff and Peel (77) confirmed that the famous allegations of high DDT levels in Antarctic samples “were overblown by more than 100-fold” because the plastic bags in which they were stored contained so much PCB (78). In 1969, Peterle (79) attempted to measure DDT in Antarctic snow and melt-water. Only one of his five filters indicated any DDT presence, but based on that single filter, he wrote that “there should be five million pounds of DDT in Antarctic snow.” (No other analyses confirmed his guess.)

PCBs occurred in most fish and wildlife samples at higher concentrations than did DDT residues. In 1970, Risebrough et al. pointed out that “PCBs are more important enzyme inducers than DDE, causing delayed breeding” (80). When ingested by birds, they were known to

cause infertility, eggshell thinning, non-hatchability, and severe physiological damage. Curley et al. (81) stated that “PCBs throw off hormonal levels in birds, affecting calcium reserves, which in turn result in thinner eggshells.”

In 1971, Risebrough and Spitzer stated that Chesapeake Bay ospreys had no trouble with DDT or DDE, but that their eggs contained PCB levels from 545 to 2,270 parts per million (82). Boyle (83) reported that Herring gulls from the Great Lakes contained an average of 2,224 ppm of PCB and “signs of eggshell thinning were found in nine of thirteen species in the region.”

Scott et al. (84) observed that “Many reports relating reproductive declines of wild birds to DDT and DDE were based on analytical procedures that did not distinguish between DDT and PCBs.” Because PCBs caused such severe effects on fish, birds, and wildlife, it was noteworthy that so great a proportion of the difficulties were later admitted to have really been PCBs, instead of DDT. Feeding birds high levels of DDT alone did not cause birds to produce those thin eggshells.

By 1971, 5,000 tons of PCBs were being produced annually in the United States. After 20 years of faulty analyses of DDT, it was finally admitted that we had been misinformed and that many individuals who *knew* the truth had deliberately concealed it to foster banning DDT!

Lichtenstein et al. (85) prepared a table revealing GLC analyses of 11 Aroclor™ plasticizers and listing various chlorinated hydrocarbon insecticides whose peaks were identical to some peaks of PCBs. Those insecticides included DDT, DDD, Lindane, heptachlor, heptachlor epoxide, aldrin, and dieldrin.

Aquatic habitats have often been seriously threatened by PCBs. Harvey (86) and (87) stated that there was “twice as much PCB as DDT in Atlantic sea water,” and Giam et al. (88) found similar levels in plankton from the Gulf of Mexico and the Caribbean. Hom et al. (89) reported that there were large amounts of PCB in the Santa Barbara Basin of southern California (near Anacapa Island), where a high “apparent DDE” concentration was reported in sediment that was deposited 12 years before any DDT or DDE existed. They explained that away by saying “we attribute the higher DDE concentration in the 1930 sediment to spurious contamination during collection, storage or analysis.”

In 1965, Anderson et al. analyzed five samples of egg-pools, which they then claimed contained high levels of DDT. Years later, after DDT had been banned, they reanalyzed those five samples and belatedly reported that three of the five samples actually contained no DDT at all, and the other two had only a fourth as much as touted in their earlier article (90). Scott et al. (84, p. 364) reminded us that those authors had led the campaign to ban DDT because of its supposedly “proven effects on eggshells,” but they later sought to implicate PCBs as having been the *real* hazard.

Sherman (91) called attention to these developments and observed that “This negated the putative data that were the basis for previous alleged sensational charges against DDT. For thirty years DDT was a scapegoat for artifacts and mimics of that pesticide.”

If the errors that were so frequently repeated by anti-DDT activists were actually unintentional, they should have corrected them. That practice of not correcting misstatements also seemed to appeal to journalists, politicians, and environmentalists. As a result, DDT was eventually placed on trial by William Ruckelshaus, the head of the Environmental Protection Agency.

In April 1972, after 7 months presiding over those EPA hearings, Judge Edmund Sweeney concluded that “DDT is not a carcinogenic hazard to man... DDT is not a mutagenic or teratogenic hazard to man... The uses of DDT under the regulations involved here do not have a deleterious effect on freshwater fish, estuarine organisms, wild birds or other wildlife. The evidence in this proceeding supports the conclusion that there is a present need for the essential uses of DDT” (92).

Incredibly, EPA Administrator Ruckelshaus (an attorney, with no knowledge of the scientific facts) never attended a single day of the seven months of hearings, and (according to his special assistant, Marshall Miller, did not even read the transcript of those vital hearings) (93). However, he overruled the judge (and ignored the hundreds of truthful scientists who had traveled to testify during the hearings). He was a strong supporter of the Environmental Defense Fund, and deliberately ignored the scientists’ 9,000 pages of testimony and personally banned DDT. Several of the serious errors and untruths in Ruckelshaus’ “Opinion and Decision” concerning DDT were exposed and were published in the *Congressional Record* by Senator Barry Goldwater in 1972 (94). The information concerning his decision was never adequately reported by the news media or by the pseudoenvironmentalists. Ruckelshaus mistakenly applied the wrong name to DDT, erroneously stated that DDE was also an insecticide, and proposed that applications of DDT be replaced by parathion (which had already killed or injured hundreds of people).

John Quarles’ June 3, 1982 affidavit to the U.S. District Court for Northern Alabama is of interest because it discussed the irresponsible Ruckelshaus ban of DDT. Quarles had served as General Council for Mr. Ruckelshaus in 1971 and 1972. He testified, “After seven months of hearings, the EPA Hearing Examiner made findings generally supportive of the position that DDT did not cause undue harm and that an adequate basis did not exist for cancelling the uses of DDT.” Quarles added that “There were no findings that DDT had caused harm or would cause harm under a specific set of circumstances or at any particular time or place.” This was of course not reported by the news media and was buried by every environmental group.

On April 26, 1979, in a letter to Allan Grant (the president of the American Farm Bureau Federation), Mr. Ruckelshaus wrote, “Decisions by the government involving the use of toxic substances are political, with a small ‘p’. Science has a role to play, but the ultimate judgement remains political, (and) the power to make this judgement has been delegated to the Administrator of the EPA.” By the time Ruckelshaus wrote that letter, he had become senior vice president of the (non-environmental) Weyerhaeuser Lumber Company! Later he took charge of the Browning-Ferris garbage disposal company and sought

to dump trash and garbage into a pleasant little stream in Apanolia Canyon, near Half Moon Bay, California. That would have destroyed irreplaceable riparian habitats and an ecological complex of great diversity.” Fortunately, local outrage against Ruckelshaus’s scheme halted his destructive actions, *that time!*

MISIDENTIFICATIONS OF DDT

Glotfelty and Caro (95) commented that “misidentifications of chlorinated hydrocarbon insecticides resulted from interference by pigment-related natural products found in photosynthetic tissues.”

Sims, plant pathologist and biochemist at University of California, discovered that certain red algae produce halogen compounds that had been misidentified as DDT metabolites. He said, “the natural halogens were unknown previously but are now known to be common in natural marine habitats. Compounds containing bromine or iodine rather than chlorine may also have registered as DDT on the gas chromatograph” (96).

Frazier et al. analyzed 34 soil samples that had been sealed in glass jars since they were collected in 1911 (97). The gas chromatograph indicated that five kinds of chlorinated hydrocarbon insecticides were in that soil, even though none of them had been in existence until 30 years after those samples were sealed. They concluded that “the apparent insecticides were actually misidentifications caused by the presence of co-extracted naturally-occurring soil components.”

Coon confirmed that material he and his colleagues at the WARF Institute earlier believed to contain DDT residues actually had none at all (98). He also reported on a gibbon that was collected in Burma in 1935 and sealed in a tight container for 30 years before being analyzed. The analysis indicated DDE in the kidneys, testes, liver and fat tissues, even though DDE did not exist anywhere on earth until 6 years after the gibbon was preserved.

Bowman et al. (99) analyzed soil that was sealed in 1940 (years before DDT) and reported “a naturally-occurring extraneous substance appearing in pre-DDT soil that gave the same chromatographic retention times on GLC as DDT (62 ppm) and DDE (35 ppb).”

Hickey sent a robin that had been collected and stored in formalin in 1938 to the WARF Institute for analysis (99). Although it was collected 7 years before DDT was present, the analysis indicated that the robin contained DDT and DDD! Dozens of other robins were similarly analyzed in Michigan and Wisconsin before it became evident that mercury in the earthworms eaten by those birds actually caused the symptoms that had been attributed to DDT poisoning, including tremors, ataxia, and death. Massive amounts of DDT fed to caged robins failed to make them ill. Even nestling robins were not sickened after they were fed only food containing DDT.

DDT SAVING FORESTS

Despite the absence of any confirmation that wild birds suffered illnesses or mortality caused by DDT, activists succeeded in halting the use of DDT on American elm trees, following which thousands of the stately trees died in

five midwestern states because of the Dutch Elm Disease that was transmitted by uncontrolled Bark Beetles. Later, millions of eastern oak trees were killed by gypsy moth larvae, which had previously been quickly controlled by a single spray of DDT. Their caterpillars defoliated more than 9 million acres of forest annually before DDT was available, but 1 pound of DDT per acre eradicated the gypsy moths from New England and all adjacent states. The National Audubon Society wrote “No damage was done to bird life, including nestlings.” USDA biologists wrote (100), “without this insecticide, there is little hope of preventing the spread throughout their potential range.” Environmentalists fought desperately to halt the use of DDT in the infested forests! Forest destruction accelerated after DDT was banned.

In 1974, a campaign was launched to spray DDT on forests in three northwestern states to halt a great Douglas-fir tussock moth outbreak. Permission to spray DDT was granted by the EPA, after 3 years of ridiculous obstinacy. The DDT spraying was then quickly completed, the pests were eradicated, and thousands of acres of Douglas-fir forests were saved. Analyses of vegetation, soil, birds, mammals, and stream inhabitants revealed no harmful effects caused by DDT. Ignoring all science, however, two local newspapers editorialized regarding the persistence of DDT. The *Vancouver Sun* and the *Lewiston Tribune* both wrote in editorials (101) that “DDT has a half-life of several thousand years.” (It had only been in existence for 30 years!) I sent each editor copies of literature citing truthful scientific data, but neither of them responded.

BIOLOGICAL MAGNIFICATION UP FOOD CHAINS

A great effort was made by environmental extremists to convince us that there was “biological magnification of DDT up food chains.” The theory was as follows: “Small crustaceans ingest DDT, and when hundreds of them are eaten by a fish, that fish might contain a higher DDT concentration than any one crustacean. If many of those fish are eaten by a duck, it might then harbor a higher concentration of DDT than any one fish. After hawks eat the ducks, they could theoretically contain tremendous amounts of DDT! The major difficulty with this theory is that it simply doesn’t work that way, where DDT is involved, because DDT is eliminated in faeces, metabolized in blood and cells, and excreted in the urine.

A typical propagandist diagram shows 0.04 ppm DDT in the zooplankton or tiny crustaceans, compared with 25 ppm in the brain of a hawk at the top of the food chain (an increase of 600 times). However, propagandists said that “DDT levels increased 10 million times.” They untruthfully sought to start the food-chain with 3 parts per trillion (ppt) of DDT in the water (*before* it entered any food chain!) They then concluded that “DDT is magnified a million or more times as it passes up the food chains.”

DDT adheres to particles in the lake bed, so the mud or sand contains a much higher DDT concentration than the water itself. As Hickey pointed out (102), “biological magnification only begins *after* the DDT gets into some living creature.” Transfers before that time are *not* “biological”, and are not any part of any food chain!

Another source of misinformation is also obvious. If *wet-weight* samples of crustaceans are compared with *dry-weight* samples of fish muscle, the dry tissues will appear to have higher concentrations of DDT because the samples are not diluted by water. Duck muscles do not contain higher concentrations of DDT than do fish muscles, if the samples are prepared and analyzed similarly (103, p. 2160).

Freed (104) proved that about 85% of the DDT in a fish enters through the gills, and very little through the diet. Davis found that “trout pass all of the blood in their body through the gills every 64 seconds, causing rapid transfer of chemicals from the water into the body tissues (105). After the fish is eaten by a duck, most of the DDT passes through the duck’s digestive tract and is eliminated.

Comparison of DDT concentrations at each step up a food chain are difficult, and the approximate levels of DDT at each step in the food chain are difficult to determine!

Animals usually store *lower* levels of DDT than were in their food. Anchovies in Monterey Bay contained no more DDT residues than the algae they fed upon (750 to 1,000 parts per billion). Sea lions were found to contain much lower concentrations of DDT than the fish upon which they fed.

Jefferies and Davis (69) reported that earthworms kept in soil saturated with dieldrin contained a lower concentration of dieldrin than the soil, and thrushes that were fed *only* on those earthworms never accumulated any higher concentration of dieldrin than the individual earthworms had contained.

POSTSCRIPT

On June 10, 1971, Philip Butler’s report for the National Academy of Sciences claimed (106), “As much as 25% of the DDT produced to date may have been transferred to the sea.” Two other panels of the NAS, headed by Kanwisher (107) and by Harvey (108), later refuted such reports. The *Washington Post* (109), however, expanded upon that untruthful claim, saying “Nearly 25% of all DDT manufactured to date is now in the world’s oceans, where it is killing baby fish.” Jacques Cousteau must have been confused by it all, for in his testimony before the Senate Commerce Committee on October 18, 1971, he stated, “For example, we now know that 25% of all DDT compounds so far produced are already in the sea, and finally they all will end up in the sea.” That false statement was repeated in *U.S. News & World Report*, (110).

So, a false estimate of how much DDT *may* have gotten into the sea (made by the man whose own EPA laboratory at Gulf Breeze, Florida, had shown that DDT and its metabolites break down very rapidly in sea water), was escalated to a solid statement by the *Washington Post* (and alleged to “kill baby fish”). It was then blown completely out of context by Cousteau and the anti-DDT press, warning that **ALL of the DDT and its metabolites ever produced will “finally . . . end up in the sea.”** The same sort of misstatements and deliberately false allegations are still being directed against many other chemicals that are of great potential benefit to mankind and the environment.

With scientists who behave like so many of those in this report, how can industry, education, conservation, and public health programs survive in America?

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INSTREAM FLOW METHODS

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Water use is typically classified into offstream use (withdrawals) and instream use. Offstream demands include industrial, commercial, agricultural, and domestic uses. Instream demands include hydropower, recreation, navigation, fish and wildlife habitat management, and assimilation of waste. Unlike offstream demands, which should be satisfied separately, instream demands in general are satisfied simultaneously. That is, the same quantity of water is utilized in instream demands. Therefore, the largest of these instream flow demands

must be satisfied. This demand may be in terms of discharge, depth of flow, and velocity. Jowett (1) classifies the methods as *historic flow regime*, *hydraulic*, and *habitat* but points out that the hydraulic methods are not usually used to assess seasonal flow requirements. A key determinant in managing instream flows is the presence of an upstream reservoir. Low flow augmentation is possible with the availability of stored water and efficient management of the reservoir is an important aspect. When a reservoir is not available, the low flows in the stream form the basis for minimum instream flow assessment. The competing offstream demands, public preference, local economy, aesthetics, regulations, water rights, and water quality form the constraints.

When there is an abundance of water, the competing needs can be met easily. When there is only a limited supply, problems emerge. Because river flows are random, it is likely that low flows will occur and proper planning should address the issue. The determination of minimum instream flow may be based on an exceedance criterion such as 90% to guarantee a relatively uninterrupted supply; it may also be based on the nature of the demand such as maintaining a minimum depth or velocity. Hydropower depends on both available flow and hydraulic head. For runoff without storage, the amount of energy that can be guaranteed over 90% (to 97%) of the time is called the firm energy. The required flow is determined as the flow that will be equaled or exceeded 90% of the time. The plot of discharge against percent time equaled or exceeded is called the *flow-duration curve*. If a reservoir is available, a set of *rule curves* is typically followed in making the releases to optimize the available storage subject to the inflow behavior (2). Navigation and recreational boating require maintenance of flow depth based on the barge and boat size. The required flow is to be determined based on the depth requirement.

The procedure given here is in general followed to determine flows that are equaled or exceeded by a certain percentage. A list of moving average values for a fixed averaging duration such as 7 d is generated. Using these moving average values, flows corresponding to various return periods such as 10 yr (90% exceedance) are determined by *frequency analysis* (3,4). The resulting flow is called 7-d 10-yr flow or 7Q10. Virginia Department of Health recommends 1-d 30-yr flow as the characteristic low flow for drought-related studies. For fish habitat, the Tennant or Montana method recommends 30% of annual mean flow as the flow providing required channel flow conditions.

Dissolved oxygen, temperature, water quality, substrate, stream toe width, velocity, flow, and depth are considered key elements in preserving fish habitat. The instream flow incremental methodology (IFIM) utilizes detailed physical habitat simulation (PHABSIM) models. In PHABSIM, for each physical variable of depth, velocity, and substrate a habitat preference curve is defined for a specified species at a specified life stage. For a flow value, the pattern of distribution of depth, velocity, cover, and substrate is analyzed over a stream reach with the aid of a hydraulic model. Using the physical variables, the values from the habitat preference curves are combined into an

index called weighted usable area (WUA). For a range of flow values, the WUA values can be obtained. Based on the weighted usable area values, a suitable set of flows can be determined for different levels of species protection and enhancement (4,5). Hardy (6) provides a comprehensive discussion of habitat modeling.

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FLOODPLAIN

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The term *floodplain* has a range of physically and ecologically based definitions, in addition to a regulatory (i.e.,

flood control) definition. From the physical perspective, floodplains are a component of the dimensions and geometry of river channels, which as a whole reflect the runoff and sediment load they carry (1). Floodplains have been defined as the relatively flat level floor formed of sediment deposits, which extends from the edge of the channel to the valley walls (2). Nanson and Croke (3) define floodplains more precisely as “the largely horizontally-bedded alluvial landform adjacent to a channel, separated from the channel by banks, and built of sediment transported by the present flow-regime.” They identify lateral point bar accretion, overbank vertical accretion, and braid-channel accretion as the dominant floodplain forming processes in their genetic floodplain classification based on stream power and sediment texture. Figure 1 is a conceptual diagram of a natural river basin that identifies the valley floor, floodplain, river corridor, and channel areas of a basin. Floodplains and riparian areas immediately adjacent to active channels have been documented as very important in sediment retention during high flows (4). Floodplains also act to dissipate bed shear stress and attenuate flood peaks contained in the primary channel. In addition, flood peaks travel downstream faster when a river corridor becomes confined laterally from its floodplain. Floodplains provide important aquatic and terrestrial habitat in addition to significant hydrologic, water quality, and aesthetic benefits (5).

The biological and ecological importance of floodplains has been well documented. Many species of aquatic organisms require floodplains to complete stages of their life history (6,7), including reproduction, rearing, and hibernation. Floodplains also provide important refugia for aquatic organisms during episodes of high flow or vulnerability to predators (8). Aquatic invertebrates and fish seek refuge from high bed shear stresses and

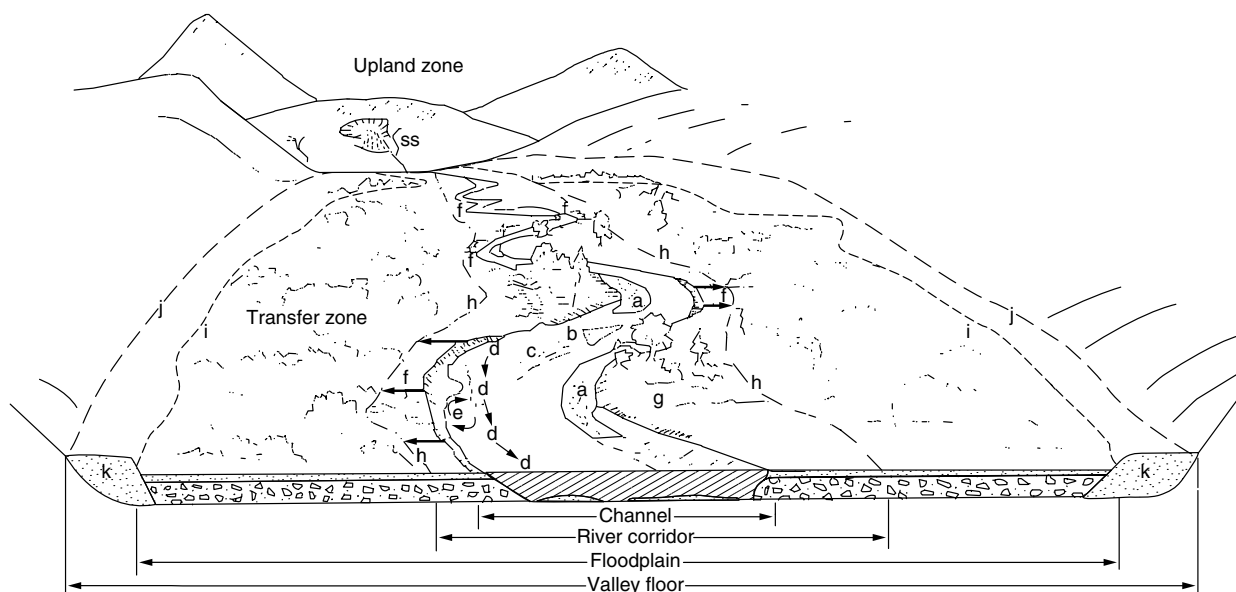


Figure 1. Generalized valley floor, floodplain, river corridor, and channel orientation (after Ref. 5). Letters refer to channel corridor characteristics: (a) point bar, (b) midchannel bar, (c) riffle, (d) pool, (e) secondary current, (f) bank erosion, (g) riparian vegetation, (h) corridor boundary, (i) floodplain boundary, (j) valley floor boundary, (k) terrace, (ss) sediment supply and storage zone.

flow velocities during floods in the hydraulically rough, slow-moving areas beyond the margins of the active channel and on the floodplain. The importance of these floodplain functions has been shown in population growth simulation models that predict increases in biomass of fish and aquatic invertebrates with increases in floodplain width (9). Floodplains have been recognized as critical elements of healthy river ecosystems (10). Without a connected and functioning floodplain, a river corridor ecosystem is incomplete. In their work on the flood pulse concept, Junk et al. (11) describe an aquatic-terrestrial transition zone that moves across the floodplain as flood waters rise. This moving transitional zone is essential to the successful recruitment, establishment, and growth of many species of vegetation, fish, and other organisms in river corridor ecosystems. Historically, floods were often viewed as destructive disturbances to floodplain ecosystems. More recently, however, the prevention of floods has been viewed as the more destructive disturbance to floodplain ecosystems.

The regulatory definition of floodplains originated because valley floors and floodplains have historically been attractive sites for human settlement (12). For thousands of years, humans have built civilizations along river corridors. In fact, the very "cradle of civilization" was built near the confluence of the Tigris and Euphrates Rivers in what is now southern Iraq. Most human development along river corridors has been established on floodplains. Floodplains are naturally flat, close to water, and have fertile soils, making them ideal for agriculture, building sites, and industry location (13). Therefore, a regulatory floodplain is a surface next to a channel that is inundated once during a given return period (14). Regulatory floodplains may or may not be related to physical floodplains. In most cases, some form of flood protection is provided on regulatory floodplains. The administration of regulatory floodplains differs from country to country. In the United States, Congress established the National Flood Insurance Program (NFIP) with the passage of the National Flood Insurance Act of 1968 to identify regulatory floodplains and administer flood protection. The Federal Emergency Management Agency (FEMA) currently administers the NFIP. In return for federally subsidized insurance, communities with access to the NFIP must adopt and enforce a floodplain management ordinance to reduce future flood risks to new construction in Special Flood Hazard Areas (SFHAs). A SFHA is an area within the 100-year floodplain (note this is not a physical reality, but rather a regulatory designation that is discussed in more detail below). Development may take place within a SFHA, provided that development complies with local floodplain management ordinances, which must meet the minimum federal requirements. Communities that participate in the NFIP must implement floodplain management measures in the development of their floodplain lands. FEMA expects these measures to constitute an overall community program of corrective and preventive measures for reducing future flood damage. In general, floodplain management measures address zoning, subdivision, or

building requirements for structures in the floodplain and special-purpose floodplain ordinances.

The 100-year flood (and floodplain) was selected by an expert panel as the standard for protection within the NFIP. The 100-year flood (or the 1% annual-chance flood) was chosen on the basis that it provides a higher level of protection (FEMA does not specify what "higher" refers to) while not imposing overly stringent requirements or the burden of excessive costs on property owners. Flood Insurance Studies (FIS), which apply detailed hydrologic and hydraulic analyses, have been performed for many of the watersheds in the United States that are currently included in the NFIP. The results of these analyses include flood insurance rate maps (FIRMs) that identify areas of flood risk and, perhaps most importantly, the boundaries of the regulatory 100-year floodway. The regulatory floodway is defined as the channel of a stream plus any adjacent floodplain areas that must be kept free of encroachment so that the entire base flood (100-year flood) discharge can be conveyed with no greater than a 1.0-foot increase in the base flood elevation.

The references listed in the Bibliography provide a thorough treatment of floodplains for those wishing to develop a deeper understanding of this broad and diverse topic.

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FISH PASSAGE FACILITIES

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A *fish passage facility* provides fish with a watered route around or through a barrier to migration in a fluvial system. Fish passage facilities have been designed for both natural and artificial barriers. Some passage facilities allow fish to propel themselves past the barrier, while others physically transport fish past the barrier. Fish passage facilities must provide passage during periods dictated by migratory requirements without injuring or unduly stressing fish. Fish passage facilities have been designed and constructed in North American, Europe, Russia, Latin American, Africa, Australia, New Zealand, China, and Japan. Depending on the region and type of application, fish passage facilities are also referred to as fishways, fish ladders, fish passes, fish bypasses, fish locks, and fish elevators.

Many fish species (e.g., Chinook salmon, American shad, striped bass) must travel upstream and downstream in river systems to complete their life cycles. The development of infrastructure (e.g., dams, water diversions, culverts at road crossings) on rivers has blocked fish passage on rivers around the world. In many cases, viable populations of some fish species have been threatened by these passage barriers. To mitigate this threat, fisheries professionals have designed and constructed fish passage facilities. Many fish passage facilities have been constructed as retrofits to existing barriers that were designed and built before fish passage was required, or before the technology was fully developed. More recently, some infrastructure on rivers has been designed in a "fish friendly" manner that integrates requirements for safe and effective fish migration in the original design (e.g., "fish-safe" turbines, pumps, spillways). As multispecies fish conservation and restoration increases in importance, fish passage facilities will be required to provide a range of hydraulic conditions (i.e., depths and velocities) throughout the year to provide passage for a range of fish species and life stages.

A variety of fish passage facility designs have been developed (see Table 1) to address a wide range of barriers to fish migration. However, as Odeh and Haro (1) note, all fish passage facilities include five common features: (1) the immediate reach of river or waterway served by the fish passage facility; (2) the entrance to the fish passage facility; (3) the fish passage facility itself; (4) the exit of the fish passage facility; and (5) the reach of river where fish exit the passage facility. An effective fish passage facility must allow fish to navigate each of these components of

Table 1. Fish Passage Facility Designs

Fish Passage Facility Type	Primary Passage Direction	Mode of Fish Movement	Energy Dissipation
Pool and weir	Upstream	Active	Turbulence in pools
Weir and orifice	Upstream	Active	Turbulence in pools
Denil	Upstream	Active	Partial reversal of flow direction
Vertical slot	Upstream	Active	Partial reversal of flow direction
Fish lock	Upstream	Passive	N/A
Fish lift	Upstream	Passive	N/A
Screen and bypass	Downstream	Active	N/A
"Nature-like"	Upstream/ downstream	Active	Turbulence from "natural bed elements"
Road crossing culverts	Upstream/ downstream	Active	Turbulence from roughness elements added to culvert

the facility safely and in a manner that meets the temporal requirements of their migratory movements.

Fish passage facilities can facilitate both upstream and downstream passage around barriers in fluvial systems. However, most facilities are designed primarily for passage in one direction or the other. Table 1 lists the most common types of fish passage facilities and summarizes for each the primary passage direction (upstream or downstream), mode of fish movement (active or passive), and mechanism of energy dissipation (e.g., turbulence). Fish passage facilities must be designed to align hydraulics and physical characteristics of the facility with the swimming capacity and behavior of the species of concern. Designers must carefully consider flow velocities, depths, turbulence, dissolved oxygen levels, temperature, noise, light, and even odor in the design of a fish passage facility.

Upstream fish passage facilities must attract migrating fish to a specified location in the river downstream of the barrier with an appropriate flow, and then move the fish actively (i.e., by encouraging them to swim) or passively (i.e., by trapping and transporting them) around the barrier. Upstream fish passage facilities for adult anadromous salmonids (e.g., salmon and trout) and clupeids (e.g., shad and alewives) have a long history in North America and Europe. Upstream fish passage facilities have also been designed for juvenile catadromous fish (e.g., American eel). The most common upstream fish passage facilities are described briefly below.

Pool and Weir. The pool and weir facility consists of a series of pools separated by weirs (Fig. 1) that allow fish to ascend by jumping or swimming from the downstream end at the river to the upstream end at the top of the barrier. Flow energy is dissipated as turbulence in pools.

Weir and Orifice. The weir and orifice facility is similar to the pool and weir except that each dividing wall between

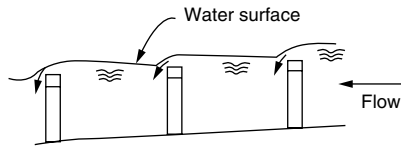


Figure 1. Pool and weir.

pools has a submerged orifice (Fig. 2). Some facilities of this type allow passage from pool to pool through an orifice or over a weir. Again, flow energy is dissipated as turbulence in pools.

Denil. The denil facility consists of a steep flume with vanes along the sides and bottom (Fig. 3). Fish ascend from the downstream end of the barrier to the upstream end by swimming through the center of the flume. The vanes in the denil partially reverse flow direction to dissipate the energy.

Vertical Slot. The vertical slot facility is similar to the pool and orifice type except that the orifice extends for the full height of the baffle between pools (Fig. 4), allowing migrating fish to pass upstream from pool to pool at any depth in the water column. Similar to the denil, the vertical slot dissipates energy by partially reversing the direction of the flow in the facility.

Fish Lock. A fish lock operates in a similar manner as a navigation lock (Fig. 5). Fish enter a chamber at the downstream end of the barrier that is flooded to raise the water surface elevation in the chamber. Fish are then passed upstream through chambers until the water surface elevation difference between the chamber and the upstream end of the barrier is small enough for the fish to pass.

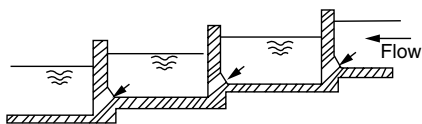


Figure 2. Weir and orifice.

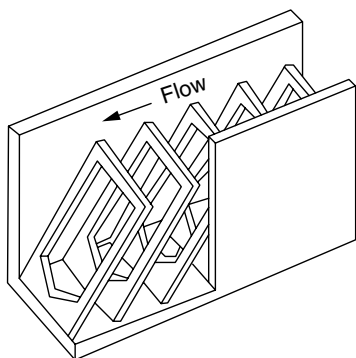


Figure 3. Denil.

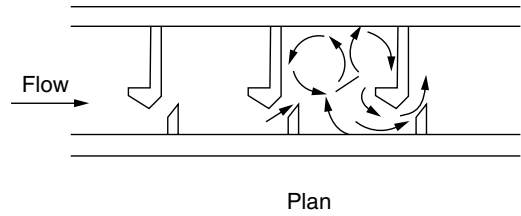


Figure 4. Vertical slot.

Fish Lift. A fish lift can include any mechanical means of transporting fish around a barrier such as tank trucks, large buckets, or other flooded containers (Fig. 6). Migrating fish must be lured into the flooded container and then provided with suitable conditions to maintain their health for the duration of the lift.

Downstream fish passage has received significantly less attention than upstream passage, and therefore the facilities designed to provide downstream passage are less developed. To date, no country has found a satisfactory solution to downstream migration problems, especially at large passage barriers (2). Downstream fish passage facilities have been developed primarily for juvenile anadromous fish. Most downstream fish passage facilities consist of some kind of screen and a bypass. The screen diverts fish away from potential harm associated with the passage barrier (e.g., turbines or agricultural diversions) and the bypass returns fish to the river downstream of the passage barrier. Bypasses must be designed to accommodate the behavior and swimming abilities of the fish species of interest. The most common screens are described briefly below.

Physical Screens. Physical screens (Fig. 7) typically divert downstream migrating fish away from potential harm using screens made of a wide variety of materials (e.g., perforated plates, metal bars, wedgewire, plastic or metal mesh). Screens are normally placed diagonally to flow and must meet maximum approach velocity (i.e., the velocity of flow passing through the screen) and mesh size criteria suitable to prevent impingement of the species of interest. This category of barriers includes drum screens, inclined plane screens, self-cleaning screens, and screens installed perpendicular to flow.

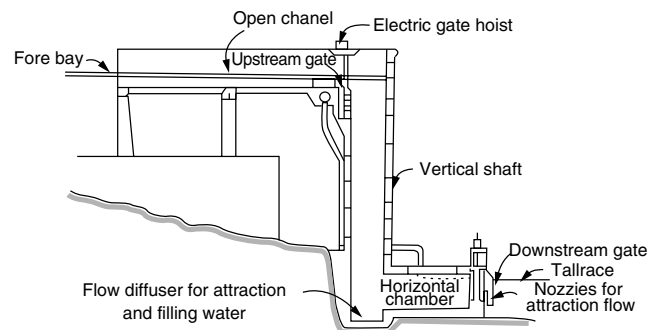


Figure 5. Fish lock.

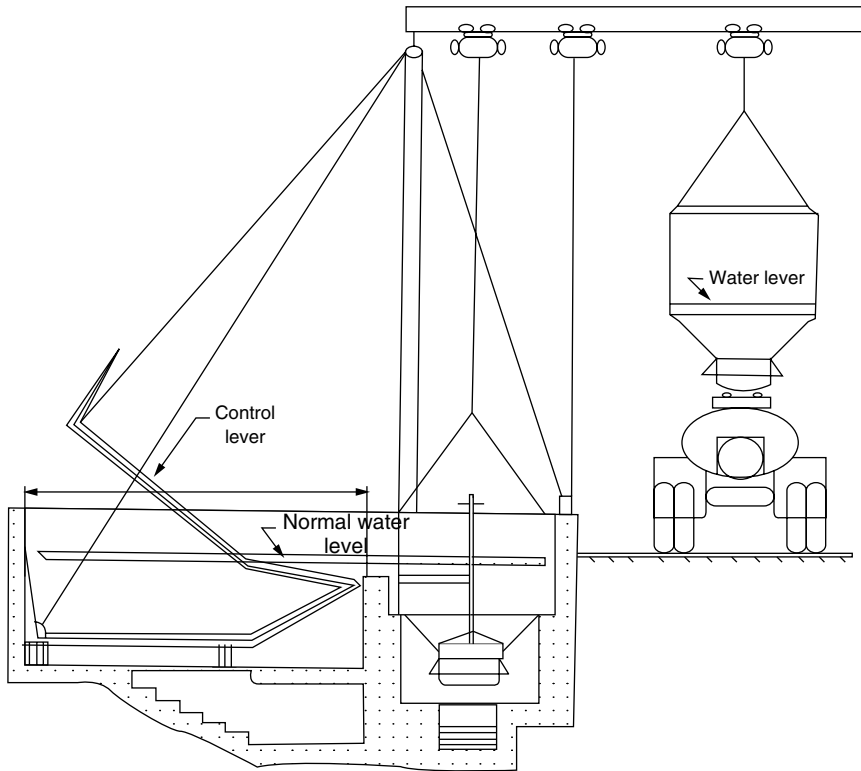


Figure 6. Fish lift.

Behavioral Screens. Behavioral screen facilities for downstream passage have been designed to take advantage of behavioral responses of the fish species of interest to attract or repel fish from a specified location. These approaches have relied on visual (e.g., air bubbles or light arrays), auditory (e.g., audio signal arrays), electrical (e.g., electric currents), and hydrodynamic (e.g., altered velocities and flow patterns) signals to elicit the desired response from the fish species of concern.

Some fish passage facilities provide both upstream and downstream passage to a variety of fish species. These facilities are typically designed for barriers with relatively small water surface elevation differences. Two of these types of facilities are described briefly below.

“Nature Like”. Nature-like fish passage facilities incorporate features of natural channels to dissipate energy and

provide suitable passage conditions for the fish species of interest (Fig. 8). Specific designs of this type include rock-ramp fishways, ramp fishways with boulders and pools, and boulder block fishways. Each of these approaches achieves upstream and downstream passage in a manner that is more aesthetically compatible with the natural stream environment.

Road Crossing Culverts. When designed to provide passage for fish under a range of flow conditions, culverts at road crossings can be considered fish passage facilities. Inappropriately designed culverts are often barriers to fish passage. New culvert designs and culvert retrofit designs for fish passage must consider the upstream and downstream migration behavior and the swimming ability of the fish species and life stages of interest. Culverts with baffles, slotted weirs, spoilers, and more natural roughness elements such as cemented boulders have been installed to provide adequate passage conditions.

A significant amount of research and development has occurred in the field of fish passage facilities. References 1–7 provide a good starting point for further investigation of this topic.

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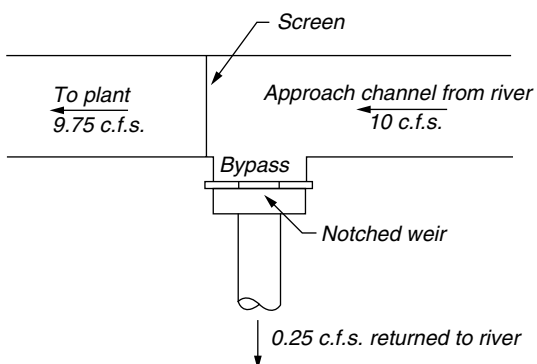


Figure 7. Physical screen.

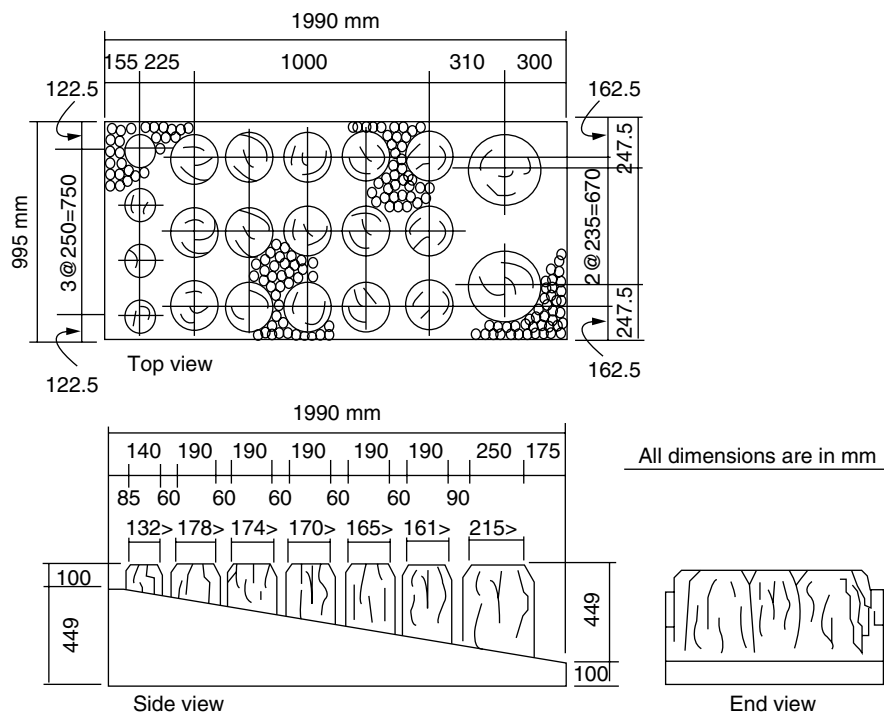


Figure 8. Nature-like fish passage facilities.

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FISHING WATERS

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The term *fishing waters* refers to all bodies of water where fish are legally captured by humans. Fishing waters can be defined and regulated internationally, nationally, and locally. The four primary categories of fishing that occur in fishing waters are commercial, recreational, subsistence, and indigenous. More than one category of fishing can occur in a given fishing water, and extractive fishing does not have to occur in all fishing waters.

International fishing waters include vast areas of open ocean outside of the zones managed under national fishing regulations. Fishing in international waters is governed by conventions enacted by the United Nations. The most recent conventions dealing with international fishing

waters are the Convention on Fishing and Conservation of Living Resources of the High Seas, and the Convention on the Law of the Sea. Nearly all of the fishing in international waters is commercial in nature.

National fishing waters include oceanic zones nearer to coasts and managed by individual nations. Each country manages fishing in these fishing waters differently. In the United States, national fishing waters are governed by The Magnuson–Stevens Fishery Conservation and Management Act. This Act was signed into law in 1976 and covers all fishing activities that occur in federal waters within the United States' 200 nautical mile limit, referred to as the Exclusive Economic Zone (EEZ). The Act also sets forth a system for the management and monitoring of the fish stocks in waters within the United States' EEZ. This system includes eight regional Fishery Management Councils to provide a forum for state, industry, and public participation in the management of American ocean fisheries. The National Marine Fisheries Service (NMFS) is the agency responsible for administering the fishery management requirements of the Act. The Fishery Management Councils develop fishery management plans for each zone that are administered by the NMFS and enforced by the U.S. Coast Guard. Most of the fishing in national waters is commercial in nature.

Local fishing waters can include near shore ocean areas, lakes, reservoirs, rivers, and streams. Again, each country manages local fishing waters differently. In the United States, the U.S. Fish and Wildlife Service together with state Departments of Fish and Game are primarily responsible for the management of fishing in local waters. These agencies are also involved in stocking programs that are used to maintain, enhance, and create fishing waters. In addition, these agencies assist in the management of subsistence and indigenous fisheries. State-issued licenses are

typically required to fish in local fishing waters, and most of the fishing in these waters is recreational in nature.

As introduced above, the primary requirement for a body of water to be defined as a fishing water is for some type of fishing to occur in that body of water. Commercial fishing is the most extensive practice that defines a body of water as a fishing water. Commercial fishing occurs in every coastal region in the United States and many different techniques and types of equipment are employed in this endeavor. Any fishing that results in the sale of the captured fish is considered commercial fishing. Recreational fishing is probably the second most extensive practice that defines a body of water as a fishing water. Fishing is considered recreational when fish are captured for pleasure rather than profit. Recreational fishing includes both “catch-and-keep” fishing and “catch-and-release” fishing. Subsistence and indigenous fishing are significantly less extensive practices that define bodies of water as fishing waters. Subsistence fishing provides food or other products for the people who catch the fish, and indigenous fishing is provided for and managed by treaties between the federal government and native peoples (in the United States).

The topic of fishing waters is large and diverse. References 1–4 provide a useful introduction to the topic.

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LAND SURFACE MODELING

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INTRODUCTION

Land surface modeling consists of developing and applying computational models that integrate complicated hydrologic and physical processes that occur at the land surface, between the land surface and the atmosphere, and within the soil column. A land surface model or macroscale hydrologic model, as these types of models are called, has two important characteristics: (a) it considers processes related to both the energy and water budgets and their interactions, and (b) it operates over relatively large spatial domains with short temporal scales. Depending on their complexity, these models generally include

representations for some or all of the following major physical processes:

- Snow and snowmelt
- Frozen soil
- Infiltration
- Surface runoff
- Subsurface runoff (e.g., flows from unsaturated and/or saturated zones)
- Evaporation and transpiration
- Vegetation dynamics
- Processes related to the energy budget (e.g., radiation, sensible heat flux, ground heat flux, etc.)

The above processes are interconnected and constrained by the available water and energy. Evapotranspiration links the energy and water aspects of a land surface model together. Soil moisture and surface temperature relate various different physical, hydrological, and biogeochemical processes to one another.

MOTIVATION

One of the main purposes of developing a land surface model is to couple it with (i.e., connect it to) atmospheric or climate models to study the influence of the atmosphere on the land surface, and vice versa, the feedbacks or impacts of the land surface (e.g., the processes listed above) on the atmosphere. Adequate representations of such two-way feedbacks (called land-atmosphere interactions) can improve the accuracy of numerical weather forecasts and climate predictions that play a dominant role in hydrologic forecasting. A land surface model (or a macroscale hydrologic model) is also a physically based tool to assess water resources and hydrologic impacts caused by climatic changes over large river basins or continents. Hydrologic applications at such large spatial scales are impossible without the development of the concept and approach of land surface modeling.

BACKGROUND

We often hear the terms “land surface model” and “hydrologic model.” Do they refer to the same thing? Strictly speaking, traditional hydrologic models do not belong to the category of land surface models, which is because hydrologic models do not describe energy budget-related processes, such as radiation and sensible and ground heat fluxes, although some may include limited energy related calculations such as that related to snow and snowmelt processes. In addition, land surface models have more detailed representations of vegetation-related processes than traditional hydrologic models. Furthermore, traditional hydrologic models mainly focus on the movement of water and its distribution at the land surface and within the vadose zone over small spatial domains with relatively large temporal scales. As atmospheric models deal with physical processes happening over much larger spatial domains at much

shorter time steps or temporal scales (e.g., weather phenomena can occur on time scales of minutes), land surface models must also have the capability to deal with hydrologic and physical processes at short temporal scales over large spatial domains for representation of heat and water exchange between the land surface and the atmosphere.

Although land surface models and traditional hydrologic models are developed for different applications, they share some common features as evident by the evolution of these models. The work by Manabe (1) was the first to include land surface impacts in a climate model. Although his land surface model (called the "Bucket" model) includes overly simplified hydrologic processes, a simple energy balance equation, and no vegetation, Manabe's pioneer work ignited many innovative and significant development in the later generation of land surface models.

The evolution of land surface models since Manabe's Bucket model can be characterized by three major phases/stages. The first phase was led by the work of Deardorff (2) (e.g., the force-restore scheme), Dickinson et al. (3,4) (e.g., the BATS model), and Sellers et al. (5,6) (e.g., the SiB model) in which much better vegetation representations and more complex energy budget calculation are considered to realistically represent the energy exchange between the land surface and the atmosphere. To date, a large number of novel models were developed with their roots linked to these three schemes [e.g., (7–15)].

The second phase was led by the work of Liang et al. (16,17) (e.g., the VIC model), Famiglietti and Wood (18), and Schaake et al. (19) in which the effects of subgrid spatial variability on water and energy budgets (e.g., surface runoff, evapotranspiration, etc.) because of heterogeneity of soil properties, topography, vegetation, and precipitation are considered using statistical-dynamical approaches. Also, this phase is marked by improved parameterizations of hydrologic processes (e.g., infiltration, subsurface runoff, etc.) that were overly simplified in the past to provide realistic representations of water and energy exchanges between the land surface and the atmosphere. The oversimplifications of hydrologic processes [e.g., (1)] have been found to lead to significant errors in water and energy budget-related calculations (e.g., soil moisture, runoff, and evaporation) [e.g., (20)]. Crossley et al. (21) and Gedney and Cox (22) also noted that inadequate representations of major hydrologic processes can limit our ability of projecting future climate change. With the advancement over the last decade, major hydrologic processes over the land surface and within the vadose zone are now more adequately represented [e.g., (23–26)] in many land surface models.

Motivated by the need of a more comprehensive representation of the carbon cycle to address climate change issues, the third phase was led by the work of Bonan (27), Sellers et al. (6), Cox et al. (28), and Dai et al. (29) in which vegetation dynamics are included to account for carbon uptake by plants and feedbacks between climate and vegetation. Pitman (30) provides a comprehensive discussion of these models designed for coupling to climate models.

As the land surface impacts hydrologic, energy, and carbon cycles through the interactions among hydrological, physical, and biogeochemical processes, it is essential that these processes be considered or treated in an **integrated** way through a unified representation of the atmosphere-land system. Land surface models have evolved substantially toward this goal. With the new generation of land surface models, it is now possible to address complex issues and quantify the impacts of surface water hydrology and vegetation on the physical processes that govern atmospheric circulation, and vice versa. For example, Xue (31) showed that degradation of the land surface can have a significant impact on the Sahelian regional climate by increasing surface air temperature and reducing precipitation, runoff, and soil moisture during the summer season. Xue et al. (32) also demonstrated the important role of the land surface for simulating more realistic monthly mean precipitation and flood areas over the United States. Entekhabi et al. (33) further showed the significant impact of soil moisture on numerical forecasting of extreme events (i.e., extreme events of drought and flood). Their study also established the role of land memory as an important source of climate predictability for the United States. Pitman et al. (34) showed the important impact of land cover change on rainfall trends in Australia that are consistent with observations.

LAND SURFACE (MACROSCALE HYDROLOGIC) MODELING FRAMEWORK

In a land surface model, processes associated with the water budget that were identified in the Introduction are connected by the Richards equation. The Richards equation can be written in one-dimensional, two-dimensional, or three-dimensional form. The one-dimensional (1-D) form is more typically used in land surface models. The change of soil moisture content throughout the soil column (i.e., from the soil surface to bedrock) with time can be described by the 1-D Richards equation as

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left(D(\theta) \frac{\partial \theta}{\partial z} \right) - \frac{\partial K(\theta)}{\partial z} - \sin k(z, t) \quad (1)$$

where θ is the volumetric soil moisture content [L^3/L^3], $D(\theta)$ is the hydraulic diffusivity [L^2/T], $K(\theta)$ is the hydraulic conductivity [L/T], $\sin k(z, t)$ is a sink term (e.g., to represent transpiration, E_t , from the root zone), and z represents the vertical direction and is assumed positive downward in Eq. (1). If $q(z, t)$ represents the flux [L/T] across a horizontal plane at depth z and time t , then the boundary condition at the land surface can be expressed as

$$q(z, t)|_{z=0} = P - R - E_o \quad (2)$$

where P is the precipitation rate, throughfall rate, or snowmelt rate [L/T]; R is the total surface runoff rate (i.e., overland flow) [L/T]; and E_o is the evaporation from bare soil [L/T]. If no flux across the lower boundary exists (i.e., the interface between soil column and bedrock at depth

$z = D$), but the drainage Q_b [L/T], then the lower boundary condition can be expressed as

$$q(z, t)|_{z=D} = Q_b \quad (3)$$

Processes associated with the energy budget as identified in the Introduction are connected to the water budget processes through evapotranspiration. Applying the energy balance equation at the land surface, we have

$$R_n = LE + SH + G \quad (4)$$

where R_n is the net radiation [$\text{EL}^{-2}\text{T}^{-1}$], LE is the evapotranspiration [$\text{EL}^{-2}\text{T}^{-1}$], SH is the sensible heat flux [$\text{EL}^{-2}\text{T}^{-1}$], and G is the ground heat flux [$\text{EL}^{-2}\text{T}^{-1}$], all of which can be expressed as a function of the surface temperature, T_s . The net radiation includes net shortwave radiation and net longwave radiation, which are expressed as

$$R_n = (1 - \alpha)S_{\text{down}} + L_{\text{down}} - \varepsilon\sigma T_s^4 \quad (5)$$

where α is the surface albedo, S_{down} and L_{down} are the downward shortwave and longwave radiation [$\text{EL}^{-2}\text{T}^{-1}$], respectively, ε is the surface emissivity, and σ is the Stefan–Boltzmann constant ($\sigma = 5.67 \times 10^{-8} \text{ W K}^{-4}\text{m}^{-2}$). Furthermore, if E represents evapotranspiration [L/T], then E and LE are related as follows:

$$LE = L_e \rho_w E \quad (6)$$

where L_e is the latent heat of vaporization [EM^{-1}] and ρ_w is the density of water [M/L^3]. Hence, the evapotranspiration links the water and energy budgets.

Theoretically speaking, one can obtain the soil moisture θ at different soil depth and the surface temperature T_s at any time t based on Equations (1)–(6), provided the initial soil moisture condition is known, because the processes governing P , R , Q_b , E , SH , and G may be related to either θ or T_s , or both, among other things. Practically, however, it is not feasible to solve for θ and T_s analytically based on Equations (1)–(6) and the initial soil moisture condition unless extreme simplifications are made/introduced.

As a result of the complexity and nonlinearity of the physical processes, the soil moisture θ and surface temperature T_s are usually solved **numerically** in a land surface model. The degree of complexity with which the processes are represented physically and/or numerically determines the degree of complexity of a land surface model. For instance, a land surface model can consider either or both infiltration excess runoff and saturation excess runoff in the surface runoff (R) generation mechanisms [e.g., (24)]. Different approaches can also be used to describe evapotranspiration (e.g., Penman–Monteith, supply-demand, mass transfer, α -formulation, etc.), with different degrees of complexity and accuracy.

To numerically solve Equations (1)–(6) (or their variations), land surface models are applied to a study domain (i.e., a study region) that is divided by grids in the horizontal plane (e.g., at the land surface), called model grids, and the vertical soil column is divided into a number of discrete layers. Therefore, each modeling unit has specified

lengths in the x - and y -horizontal directions associated to the model grid, and a depth in the vertical or along the soil column. The size of a model grid (i.e., its lengths in the x - and y -directions) determines the spatial resolution at which the land surface model is applied. Finer (coarser) spatial resolutions refer to smaller (larger) sizes of the horizontal modeling grids. The vertical resolution is normally defined by the depth of each vertical layer in the soil column. Higher vertical resolution implies smaller soil depth for each soil layer. Within a horizontal modeling grid, multiple land cover types are allowed. A study area may contain several modeling grids depending on its size. The vertical soil layers include both the saturated and unsaturated zones, separated by the groundwater table.

Most of the current generation land surface models lump the unsaturated and saturated zones together and treat the entire soil column as an unsaturated zone. As an exception, Liang et al. (35) presented a new approach to dynamically represent the movement of the groundwater table within the soil column in a land surface model to consider the impacts of surface water and groundwater interactions on energy and water exchanges. Their results showed the important impacts of such interactions on the partitioning of water budget components, and suggested more attention on issues in this area. Although the idea of investigating the impacts of surface and groundwater interactions on the land-atmosphere system has not yet received much attention, the dynamical representation [e.g., (35)] of groundwater table in land surface models may provide new insights about the water cycle as a whole. For example, is groundwater discharge a significant source of fresh water recycling to the estuaries and the oceans? Can groundwater recharge and discharge processes provide feedback mechanisms that affect climate?

Typical land surface models may, for example, have one vegetation layer, and three vertical soil layers in the soil column (i.e., from the land surface to the bedrock). The three soil layers are normally divided into a top thin soil layer (e.g., 5–10 cm deep), an upper soil layer (e.g., 20–200 cm deep), and a lower soil layer (e.g., from 50 cm to a few meters). The top thin layer is used to capture the quick bare soil evaporation following small summer rainfall events and to facilitate applications using remote sensing information on near-surface soil moisture. The upper soil layer is used to represent the dynamic response of the soil to rainfall events, and the lower layer is used to characterize the seasonal soil moisture behavior.

As a result of the characteristics and intended applications, land surface models typically have large grid sizes (e.g., from a few km^2 to hundreds of km^2 or larger). Therefore, it is very important to consider subgrid spatial variability (i.e., variations at spatial scales finer than the modeling grid) in soil properties, vegetation properties, topography, precipitation, etc., as these variables relate nonlinearly to the physical and hydrologic processes, such as the infiltration, runoff generation, evapotranspiration, etc., that are represented and solved only at the scale of the model grid. This problem is very challenging, and it is not fully addressed yet, although significant progress has been made in recent years.

After discretizing the study domain into modeling grids and vertical soil layers, Equations (1)–(6), together with the initial soil moisture conditions and the representations (or parameterizations) of each relevant physical and hydrologic process (e.g., surface runoff, subsurface runoff, evapotranspiration, etc.), T_s and θ are numerically solved at each model grid and for each soil layer to obtain R , Q_b , E , SH , and G , for example, at each time step t within a study period.

INPUTS AND OUTPUTS

Model Inputs

A typical land surface model requires four types of input information: (a) meteorological data, (b) vegetation data, (c) soil property data, and (d) model parameters.

Meteorological data may include precipitation, air temperature, humidity, surface pressure, wind speed, and downward shortwave and longwave radiation, or a subset of them, when the land surface model is executed offline, i.e., when the land surface model is not coupled with an atmospheric model. In the offline mode, some relationships (both empirical and semitheoretical) can be used to calculate humidity, surface pressure, and radiation, so that only the time series of precipitation, air temperature, and wind speed are needed. If the land surface model is coupled with an atmospheric model, the meteorological information listed above is provided by the atmospheric model instead.

Vegetation data generally include vegetation type, vegetation fractional coverage, leaf area index (LAI) or normalized difference vegetation index (NDVI), root distribution in the soil column (usually difficult to obtain), minimum stomatal resistance, displacement height, and roughness length, and their seasonal dependence. Depending on the complexity of the parameterizations, some models may require more information, such as green leaf fraction, stem and dead-matter area index, and light sensitivity factor, whereas others require less.

Soil property data usually include soil type, soil porosity, wilting point, field capacity, saturated hydraulic conductivity, soil water potential at saturation, soil heat capacity, and soil thermal conductivity. Similar to vegetation data, some models may require more information (e.g., the Clapp–Hornberger B parameter, soil temperature at the model lower boundary, etc.) than others, depending on the level of model complexity.

Most land surface models have some model parameters that cannot be precisely estimated because of observational limits or the absence of direct relationships with available observations. These model parameters are usually estimated based on model calibration (see the discussion on calibration in this encyclopedia) using observed time series, such as streamflow, evapotranspiration, etc. Methods for estimating model parameters through calibration can be found in the literature [e.g., (36–39)].

Model Outputs

A typical land surface model provides the following time series as output:

- snow water equivalent, depth of snowpack, temperature of snowpack;
- surface runoff, subsurface runoff, soil moisture for each soil layer;
- latent, sensible, and ground heat fluxes, and net radiation;
- surface temperature, and soil temperature of each soil layer.

Land surface models that consider surface and groundwater interactions can also provide additional information, such as time series of groundwater table depth, recharge, and discharge. Land surface models that are coupled with a river network routing scheme can also yield time series of streamflow as outputs.

REMARKS

In the past decade, the scientific community has made significant progress in the development of land surface models over three major phases/stages. Land surface models will continue to evolve to include more complete and important processes, such as surface and groundwater interactions, sediment transport, and biogeochemical processes (e.g., for studying carbon and nitrogen exchange), and to improve the representations (or parameterizations) of subgrid spatial variability associated with hydrologic and other physical processes of the integrated atmosphere-vegetation-land-soil system. It is anticipated that with further advancement of land surface models, processes such as floods, droughts, desertification, and the biogeochemical cycles (e.g., carbon, nitrogen, etc.) can be quantitatively and more realistically investigated, and through new discovery, improved understanding of the fundamental laws that control these phenomena may be revealed. Moreover, it is anticipated that new interdisciplinary fields will emerge that integrate atmospheric science, hydrology, public health, ecology, biology, etc. to investigate and predict potential epidemic outbreaks of environmental and/or waterborne diseases that result from integrated effects of natural extremes (e.g., flood and drought), human activity, land cover, land use change, etc.

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AGRICULTURAL WATER

ANIMAL FARMING OPERATIONS: GROUNDWATER QUALITY ISSUES

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Animal farming operations (AFOs) include feedlots, dairies, cattle farms, hog farms, chicken farms, and other facilities that raise or maintain a significant number of animals either seasonally or year-round. With respect to groundwater quality, the fate, reuse, and disposal of animal manure produced at AFOs is of critical concern.

Groundwater benefits a wide variety of uses including production as drinking water for domestic/municipal use, production as irrigation water for agricultural/municipal uses, ecosystem support, and providing base flow in rivers and streams that is vital to the ecological health of these surface waters. AFO related groundwater quality issues result from potentially detrimental impacts during the collection, transport, storage, and land application of animal manure and animal wastes.

Potential groundwater pollutants originating from AFOs are (1) salts and nutrients, (2) bacteria that indicate the presence of pathogens, and (3) organic or inorganic chemicals specifically associated with the management of AFOs. Table 1 lists water quality standards and goals for some of these pollutants, given various groundwater uses. As in other nonanimal farms, any crop production areas that are part of the AFO are also subject to application of pesticides and commercial fertilizers, which may affect groundwater quality.

Animal manure is physically characterized by its weight, volume, moisture content, and solids content. Key chemical properties of manure include nitrogen, phosphorus, and potassium content, 5-day biological oxygen demand (BOD_5), and chemical oxygen demand (COD). Nitrogen content is differentiated by organic nitrogen, ammonium-nitrogen, and nitrate-nitrogen content. Typical values of the chemical composition of animal manure can be found in USDA-NRCS (1) or in the American Society of Agricultural Engineers (ASAE) Standards (X384.2: Manure Production and Characteristics) (2). The specific composition of freshly excreted animal manure varies over a wide range depending on the animal species, the age/production stage of the animal, the diet of the animal, and the effects of weather, season, and animal care. The specific operation of the manure collection, transport, and storage facilities also greatly affect animal manure and waste water composition due to mixing (including mixing with nonmanure wastewaters collected within the AFO facility), separation of solids and liquids, waste digestion, aging, and any incidental or constructed waste treatment processes.

Animal manure is subject to leaching into the subsurface anywhere within the AFO and its land application area, except in concrete-lined or otherwise isolated areas of the operation. This includes unlined feedlots, open corrals, manure solids storage areas,

manure and wastewater storage lagoons, and the fields that receive manure from land application in either liquid or solid form or as a sludge amendment.

The leaching of chemicals and pathogens into groundwater is greatly affected by the timing and amount of recharge, which may originate from precipitation, irrigation, application of manure slurry, or from ponding of wastewater (e.g., in storage lagoons). Soil texture, soil permeability, organic matter content, cation exchange capacity, and soil temperature further influence the fate of some of the salts, nutrients, chemicals, and pathogens released from manure excretion and manure land applications.

NITROGEN

Animal manure contains predominantly organic nitrogen, N_{org} , and ammonium nitrogen, NH_4-N . Organic nitrogen is the dominant form of nitrogen in manure solids (particularly if they are separated from the liquid and dried). The liquid portion of animal manure is rich in both organic and ammonia nitrogen.

Organic nitrogen, if applied to land, is highly sorptive and has very limited mobility: significant amounts of organic nitrogen are usually found only in the top 5–30 cm, depending on land management practices. Over time, organic nitrogen will be converted to ammonium ion (NH_4^+), a process referred to as mineralization. The mineralization rate at which organic nitrogen converts to ammonium depends on the manure composition, moisture conditions, and temperature. After application onto soils or incorporation into soils, higher temperature under sufficiently moist conditions leads to rapidly increasing mineralization rates.

Ammonium is subject to two major environmental pathways: volatilization as ammonia gas (NH_3) and microbially facilitated oxidation to nitrate nitrogen, NO_3-N . Ammonia volatilization may account for a significant amount of the nitrogen loss from an AFO and is a major environmental pathway of the total nitrogen excreted in AFOs. High temperature and exposure of large surface areas of manure to wind or air movement (e.g., during the land application process) facilitate volatilization. Once incorporated into the soil, ammonium volatilization is limited due to strong sorption of the positively charged ammonium ion to soil particles. Ammonium mobility in soils is higher than that of organic nitrogen but still sufficiently limited to prevent significant ammonium movement to depths of more than 1–2 m. In sandy soils with limited cation exchange capacity, ammonium mobility can be enhanced if soil/groundwater redox conditions are anoxic.

Under aerated, oxic conditions, microbial oxidation of ammonium in soils is responsible for the conversion of ammonium to nitrate (NO_3^-). This process is referred to as “nitrification.” The time frame needed for complete nitrification of soil ammonium varies from hours or days under warm, moist conditions to months under near freezing conditions.

Table 1. Selected Water Quality Standards and Guidelines for Various Uses

Contaminant	Drinking Water	Livestock Water Supply ^c	Industrial Uses ^c
Nitrate (as N)	10 mg/L ^a	100 mg/L	Petroleum: 2 mg/L
Selenium	0.045 mg/L ^a		
Total coliform bacteria	No more than one coliform-positive sample/month (or no more than 5% coliform-positive for systems that analyze more than 40 samples/month) ^a	<200 to <1,000,000	
Fecal coliform bacteria	0 ^a	<1 to <10	
Iron	0.3 mg/L ^b	—	
Manganese	0.05 mg/L ^b	—	
Chloride	250 mg/L ^b	—	
Sulfate	250 mg/L ^b	<250 mg/L to <2,000 mg/L	
Total dissolved solids	500 mg/L ^b	<500 mg/L to <3,000 mg/L	Textile: 150 mg/L Cooling water: 1000 mg/L Paper: 1000 mg/L Chemical: 2500 mg/L

^aPrimary maximum contaminant levels, U.S. Environmental Protection Agency.

^bSecondary maximum contaminant levels, U.S. Environmental Protection Agency.

^cReference 1 (available at <http://www.nrcs.usda.gov>).

Nitrate, a negatively charged ion, has no sorptive capacity and is highly mobile. It can be readily leached from shallow soils and contaminates groundwater. Under anoxic conditions, however, nitrate is reduced to harmless nitrogen gas (“denitrification”). Denitrification is significant in heavy soils and in fine-grained unsaturated sediments and rocks that are subject to anoxic conditions. Reducing conditions in groundwater also lead to significant denitrification.

Of concern to groundwater is the concentration of nitrate in the recharge to groundwater. The potential nitrate concentration is difficult to estimate because the amount of volatilization and denitrification that occurs after land application of manure is typically not known with a high degree of accuracy. Where denitrification and volatilization in the soil and unsaturated zone above the water table are not significant processes, the potential concentration of nitrate in recharge water can be obtained from estimated annual nitrogen leaching losses and from estimated amounts of annual recharge:

$$\text{NO}_3(\text{mg/L}) = \frac{\text{annual N leaching loss (kg/ha)}}{\text{annual recharge (mm)}} \times 100 \times 4.42$$

For example, leaching losses of 50 kg/ha nitrate-nitrogen per year in a location with 250 mm annual recharge potentially leads to a nitrate concentration of 89 mg/L in recharge water. The limit in drinking water set by the U.S. Environmental Protection Agency is 45 mg/L.

PHOSPHORUS

Phosphorus is not a concern with respect to the use of groundwater as drinking or irrigation water. It is only a significant concern, where groundwater

discharges into streams or lakes. Total phosphorus levels above 0.05–0.1 mg/L potentially lead to significant eutrophication of these surface waterbodies.

Manure contains significant amounts of phosphorus. However, phosphorus is strongly sorbed to soil particles and subject to plant uptake, when manure is applied to crops. On sandy soils with limited cation exchange capacity and on cracked clay soils, repeated applications of manure may result in some phosphorus leaching to shallow groundwater. If shallow groundwater is collected in tile drains, drain ditches, or if it discharges into nearby streams, elevated phosphorus concentration may become a significant concern.

SALINITY

Solid and liquid manure are rich in total salinity (determined by the amount of total dissolved solids), which is readily leachable through soil and into groundwater. Crops grown on fields that are fertilized with manure may consume a limited amount of that salinity. Salinity leaching out of the root zone into groundwater is not subject to any degradation or sorption.

PATHOGENS

Pathogens are ubiquitous in animal waste, but soils provide a significant buffer against groundwater pollution. In the subsurface, pathogens become entrained in dead-end pores, are collected via filtration onto grain surfaces, or chemically interact with the grain surfaces of the soil/aquifer material. Over time, pathogens die off (“inactivation”). However, where soils and shallow groundwater sediments have large pores (sandy soils, sandy and gravelly aquifer sediments, highly fractured rocks, and finely textured soils with macropores), pathogens travel

significant distances in relatively short periods of time. Frequent detections of total coliform concentrations in shallow groundwater underneath AFOs are the result of limited pathogen filtration capacity in the soil or unsaturated zone that overlies the groundwater. Shallow wells, often small domestic supply wells, in the immediate vicinity of the AFO are most likely to be affected.

PHARMACEUTICALS (ANTIBIOTICS AND HORMONES)

Only recently, analytical methods have been established that are sensitive enough to detect these pharmaceutical compounds in water resources. They occur in significant concentrations in the inflow to and discharge from wastewater treatment plants, in surface water, and they are also present in animal manure. Research on the fate of these organic compounds in the subsurface is still in its infancy.

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AQUACULTURE TECHNOLOGY FOR PRODUCERS

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INTRODUCTION

Aquaculture is the fastest growing sector of agriculture in many parts of the world because of the healthy nature of seafood, the increasing demands on protein worldwide, and the reduction of ocean catch. According to FAO statistics (1), aquaculture's contribution to global supplies of fish, crustaceans, and molluscs continues to grow, increasing from 3.9% of total production by weight in 1970 to 27.3% in 2000. Worldwide, the aquaculture sector has increased at a rate that exceeds that of any other agriculture sector, with an average compounded rate of 9.2% per year since 1970 compared with only 1.4% for capture fisheries and 2.8% for terrestrial farmed meat production systems. In 2000, the reported total aquaculture production was 45.7 million tons by weight and \$56.5 billion by value with 10.1 million tons and \$5.6 billion of the value coming from aquatic plants. More than half of the total world aquaculture production in 2000 was finfish, although growth in the other major species

groups continues to be rapid with no apparent slowdown in production to date.

In the United States, the National Aquaculture Act of 1980 established a national policy "to encourage the development of aquaculture," which was then followed by the development of the *Aquaculture Research and Development Strategic Plan* by the Joint Subcommittee on Aquaculture. To develop a globally competitive U.S. aquaculture industry, the plan calls for (1) improving the efficiency of U.S. aquaculture production, (2) improving aquaculture production systems, and (3) improving the sustainability and environmental compatibility of aquaculture production.

The major limitations to the development of aquaculture in many regions are availability of water resources and environmental impact. Water is critical to aquatic animals, for it both serves as the medium that supplies the required oxygen and creates the physical environment that allows for movement of the animals. As world population has grown and industries have increased their production to meet the demands of the growing population, water has become a scarce resource that has had many conflicting demands placed upon it. Consequently, the major strategy for increasing aquaculture production with this limited water resource is production intensification. As fish biomass density increases, however, the amount of oxygen dissolved naturally in water becomes limited, thus necessitating supplemental aeration and oxygenation. In addition, the increased use of artificial feed as a means to developing a high density culture system leads to water pollution that stems from the facts that aquatic animals cannot completely utilize the feed and that the animals will excrete other metabolic products such as ammonia that are toxic to the aquatic animals at elevated levels. Internally, the waste produced deteriorates the water quality and makes the water unsuitable for fish to live in, while externally, discharge of the polluted water, if not properly managed, may lead to adverse effects on the environment.

Water uses and environmental impacts of different aquaculture operations can vary significantly because of the extremely diverse natures of the aquaculture systems and their associated technologies. In contrast to terrestrial farming systems, where the bulk of global production is based on a limited number of animal and plant species, more than 210 different farmed aquatic animal and plant species were reported in 2000 (1). This great diversity reflects the large number of aquatic species that are readily adaptable to the wide range of production systems and conditions present in the different countries and regions of the world. Here, we limit our scope to major freshwater systems for finfish production, namely, ponds, raceways, recirculating systems, and cage culture, as these systems are predominant in production aquaculture.

POND SYSTEMS

Description

Ponds account for the majority of aquaculture production worldwide (2). In the simplest terms, a pond is an earthen

or concrete impoundment that holds water. Water is generally not passed through the system and therefore water is normally only required to fill the pond and replace that which is lost to evaporation and seepage (2). Pond culture is practiced worldwide with it being the predominant system employed in Asia (3), where it totals over 287,400 tons of fish processed. Pond production of channel catfish is the largest single aquaculture enterprise in the United States (4).

Regardless of the species cultured, ponds can be classified by construction method, by location, and by use (2). Typical construction classifications are levee ponds, watershed ponds, hybrid ponds, marsh ponds, beach ponds, and intertidal ponds. Levee ponds are constructed by removing the soil from the area that will become the pond bottom and require a source of pumped water to fill and maintain the pond (5). Watershed ponds are built in hilly areas through the damming of a watercourse and obtain their water from the surrounding watershed. Hybrid ponds usually consist of two or three levees across a shallow drainage basin, which provides the majority of water for the pond, but they also require a source of pumped water to meet their needs (5). Marsh ponds in coastal areas are constructed with levees above ground level. Water salinity ranges from fresh to saline and filling and draining are usually done by pumping, although some marsh ponds have weirs to allow tidal waters to move in and out (2). Beach ponds are more prominent in Asia and are constructed on the shore in sandy locations where seepage is often a problem. Concrete or plastic may be used to avoid seepage problems and water ranging in salinity from brackish to saline is usually pumped with some ponds having weirs to allow for tidal flushing (2). Intertidal ponds are located just off shore in a location where a cove or bay may be blocked off with either a levee or net. Levees are often constructed out of coral rubble or rocks. The levee is designed so as to confine the cultured species while allowing for water exchange through tidal fluctuation (2). Examples of use classifications are broodstock pond, fingerling pond, and food-fish production pond. Each of the use classifications may be employed with any of the aforementioned construction classifications.

Pond Effluent and Its Constituents

Water is discharged from ponds for several reasons. First, unintentional discharge may occur due to rain if inadequate storage capacity is provided. Second, ponds may be drained either completely or partially during harvesting activities. Lastly, water may be discharged during flushing of the pond in an attempt to improve water quality or as a disease treatment (5). Effluent volume is a factor of pond use as pond drainings vary with use. In the U.S. channel catfish industry, broodstock ponds are drained every 1–5 years while fingerling ponds are drained annually and food-fish production ponds vary between 1 and 20 years (5).

Constituents of aquaculture ponds that may affect the receiving water body include solids, organic matter, nutrients, pH, dissolved oxygen, and toxic chemicals (5). Suspended solids (mainly phytoplankton cells) may be

deposited in the receiving water and possibly lead to changes in benthic invertebrate and fish populations due to alterations in the biological oxygen demand (BOD). However, the BOD in aquaculture effluents is usually relatively low and is expressed over a longer time period than most wastewaters (6).

Properly managed pond cultures operate within the limits imposed by natural processes within the pond and make it possible to operate them with minimal impact on the environment (5). The fate of nutrients within a pond is of extreme importance to fish culture. In fact, simple mass balances show that with the above noted limited water exchange in ponds, concentrations of waste products would build to levels that could kill fish.

Waste products such as nitrogen and phosphorus generated by fish can be estimated by calculating the quantity of nutrients added to the pond through feed and so on, then subtracting the amount recovered when the fish are harvested. Unfortunately, organic matter is more difficult to calculate. Carbon may be retained by the fish, respired as carbon dioxide, or lost in waste products. In 1988, empirical values for organic matter generation in fecal solids were determined by Henderson and Bromage (7) to be between 0.3 and 0.7 kg/kg of feed. Two to four times more organic matter is produced within a pond due to phytoplankton growth from inorganic nutrients in fish waste than would be calculated solely as fish waste (8).

Luckily, natural biological, chemical, and physical processes known in limnology and waste water engineering take place within the pond, which remove much of the waste products from the water (5). In fact, after several years of production, catfish ponds average 6 mg/L total nitrogen, 0.5 mg/L total phosphorus, and 75 mg/L organic matter, which is 25%, 10%, and 10%, respectively of the expected values calculated by a mass balance over just a single year (5). Nutrient and organic levels never reach the levels predicted by simple mass balance calculations due to the following processes: nitrogen is lost from the water as a gas through denitrification and ammonia volatilization and as particulate organic matter settles to the bottom and decomposes; phosphorus is lost to pond soils as precipitates of calcium phosphate and particulate organic phosphorus; and additionally, microbial processes are continuously oxidizing organic matter within the pond (5).

Levels of dissolved oxygen, pH, or toxic chemicals within aquaculture pond water have been shown to have little effect on the receiving water body (5). In a stream study, dissolved oxygen concentrations were found to be higher downstream from catfish ponds compared to upstream levels (9). Pond water pH fluctuations beyond equilibrium levels of 6.5–8.5 are due to biological activity of phytoplankton, which usually does not coincide with discharge (10). Equilibrium pH will be quickly reestablished due to dilution, reactions with the receiving waters buffer system, and/or reaeration that will equalize dissolved carbon dioxide concentrations and moderate the pH (5). Pesticides and therapeutants used in pond culture must be EPA-approved chemicals and are used infrequently due to the expense of treating large volumes of water. Additionally, long residence times and intense

microbial activity within the pond allow chemicals to be dissipated prior to discharge (5).

Pond Operation and Management

It is expected that regulatory programs in several nations will be implemented based on best management practices (BMPs) (11). Effluent management within the context of pond aquaculture poses a difficult challenge for monitoring and treatment due to infrequent discharges and high volumes of a dilute nature. Recommendations on BMPs that are being made to the pond-raised catfish industry (5) include reducing water exchange rates, providing storage within the ponds, using high-quality feeds and efficient feeding practices, managing assimilative capacity within the pond, providing adequate aeration and circulation, positioning mechanical aerators to minimize erosion, allowing solids to settle before discharging water, reusing water that is drained from ponds, and using effluents to irrigate terrestrial crops. In terms of pond construction and renovation practices, recommended practices include optimizing the ratio of watershed to pond area, protecting against erosion, and providing sedimentation basins.

RACEWAY SYSTEM

Description

Flow-through (FT) aquaculture systems can be defined as those production systems where water enters and leaves the system without being recycled for further use. These systems are characterized based on the type of culture unit employed and the pattern of water flow through the system. Culture units that have been used include ponds, tanks, and/or flumes called raceways. The water flow, in turn, is classified as either a single pass system or a serial reuse system in which water flows, respectively, through a single culture unit or several culture units before being discharged.

Flow-through systems are typically energetically passive systems, where water flows from the farm inlet to the outlet via gravity and aeration/reaeration occurs as a result of hydraulic water cascades before and after the culture units. Much information is available concerning FT production system design, construction, and use (12–16). Currently, the most prominent flow-through system is a serial reuse raceway system constructed of concrete.

Flow-through production systems offer several production advantages resulting from the design. Typical raceway culture units possess length and width ratios of 7:1 with depths generally less than 1 m. These dimensions facilitate grading and harvesting operations as well as the easy observation of fish (17). The continuous flow of clean water transports waste metabolites out of the culture units and provides dissolved oxygen needed for life support. In most U.S. raceway operations, there is typically a quiescent zone at the end of the raceway for solids capture. The quiescent zone facilitates the settling of fecal particles as no fish is present to create disturbance. The solids accumulated in the quiescent zone are removed periodically and stored in an offline settling basin.

Aquaculture production using FT raceway systems is common worldwide for many species of fish. In the United States, FT systems are the second most commonly used aquaculture production system (18). Warm, cool, and cold water species such as catfish *Ictalurus* sp., yellow perch *Perca flavescens*, and rainbow trout *Oncorhynchus mykiss*, respectively, are cultured in FT systems. However, coldwater fish production from government hatcheries and the commercial production of salmonids, particularly rainbow trout, are by far the most common uses of flow-through aquaculture systems.

Water Use

Because no water recycling occurs in FT systems, substantial quantities of water are utilized—more than any other type of aquaculture production system. In particular, the culture of coldwater fish, which are naturally more susceptible to poor water quality conditions and require higher dissolved oxygen concentrations than warm and cool water species, requires substantial water resources. However, for all practical purposes, FT system water use is nonconsumptive and does not reduce water availability for other uses.

Depending on farm location and the available water resources, there is a wide range of FT total farm flow rates, which are beneficial in understanding system water use. For example, in the United States, rainbow trout farm total flow rates can range from less than 1699 L/min to 5097×10^2 L/min for the largest farm. On a per unit production basis, typical FT systems are able to produce 6 kg/yr of fish per 1 L/min water flow (19). Based on these values, total water use for FT aquaculture trout production in the United States can be estimated. In the United States approximately 32 million kg of trout were produced in 2000 (20). This corresponds to a nonconsumptive cumulative water flow rate of 5.3×10^6 L/min, which is 15.5% of the consumptive water flow rate required to supply New York City's 9 million people (34.1×10^6 L/min). Clearly, the magnitude of the FT flow rate per production is directly related to the quantity of effluent and has important implications on subsequent FT system environmental impacts and effluent treatments.

Environmental Impact

Environmental regulations controlling FT effluent discharge have been established in many regions (21–23). Potential flow-through effluent pollutants can be grouped into three categories: (1) pathogenic bacteria or parasites, (2) therapeutic chemicals and drugs, and (3) metabolic products and food wastes (17). Regulations primarily target metabolic and food wastes, although recent regulations have begun to address the pathogenic and therapeutic components even though evidence of the environmental impact of the first two is limited (17).

Extensive work has been done to characterize FT effluents and the associated environmental impacts due to metabolic and food wastes. Flow-through effluents contain solid and dissolved organic and nutrient pollutants. Organic pollutants are derived from feces and uneaten feed and are described by biological and chemical oxygen

demands, while the nutrient pollutants, nitrogen and phosphorus, are excreted from fish gills and urine. Flow-through effluent characterizations for locations across the world have been summarized in several reviews (17,24). These studies showed that effects from FT effluent discharge are site specific and range from insignificant to heavily impacted, although heavily impacted water bodies are few in the United States (25). The general insignificant impact of FT effluents on receiving waters has been linked to the low pollutant concentrations resulting from high system flow rates. A European study compared FT effluents to domestic wastewater and showed that total suspended solids (TSS), total phosphorus (TP), total nitrogen (TN), and biological oxygen demand (BOD) concentrations were far less, 3% or less, than the same components in medium strength domestic wastewater (26).

Effluent Treatment

The treatment of FT effluents is difficult due to the dilute concentrations, high flow rates, and the high costs of traditional treatment technologies (26). Currently, strategies to treat FT effluent discharge include implementing best management practices for solid waste control (23); optimizing dietary P requirements to reduce excess P excretions (27); and/or utilizing mechanical screen filters, submerged biofilters, or other effluent treatment processes (22,24). These strategies have effectively lowered pollutant discharges from many FT systems. However, the scale-up of the effluent treatment options for applications in large facilities is still problematic with applications of many of the best available technologies to FT effluents for solids and nutrient control being limited by treatment efficiencies and/or inhibitory treatment costs resulting from treatment process operations and/or sizing considerations (28).

NET PEN SYSTEM

Description

Net pens are commonly located in coastal waters although they may also be found in open ocean or freshwater ponds and lakes. Coastal net pens are usually a net bag suspended below a floating rigid top frame. Typical net pens of this type are 10–20 m per side at the surface and 10 or more meters deep. Open ocean net pens are designed with a more rigid structure and can be as large as thousands of cubic meters in volume. Freshwater net pens are generally relatively small rigid structures covered by netting or wire mesh.

Until recently, freshwater net pens were the most common form of aquaculture being utilized throughout the world. However, this culturing technique is no longer the dominant form for freshwater aquaculture, although freshwater net pen culturing is becoming increasingly popular in China. Today, its most dominant application is coastal aquaculture with a trend toward moving further offshore. Marine net pen aquaculture has become progressively more intensive due to new technologies,

expansion of suitable sites, improved feeds, increased understanding of the cultured species biology, and increased water quality within farming systems. In some countries, marine net pen culture is the dominant culture method. Atlantic salmon are currently being cultured in net pens and cages in Canada, Chile, Ireland, Japan, Scotland, Norway, and the United States. Other species currently being cultured in net pens throughout the world include sea bream and sea bass in the Mediterranean Sea, yellowtail and red sea bream in Japan, and tuna and cobia culture is growing.

Objections to net pen culture include degradation of the environment, disease transmission, antibiotic use, and interaction of escapees with wild populations. Law suits have been filed for odors, excessive noise, and even visual pollution. Environmental impacts from intensive aquaculture include hypernutrification (eutrophication), benthic enrichment, increased biological oxygen demand, and changes in the bacterial flora. Serious impacts from net pen culture occurred in Japan in the 1990s, resulting in changes in culture practices, and Taiwan is planning for sustainable marine aquaculture as it moves toward cage culture.

Water Quality

Water quality may be affected by net pen culture due to waste metabolites released by the fish and dissolution of nutrients from the feed and feces. Impacts are affected by bottom topography, geography, and hydrographic conditions and tend to be more severe if the location chosen has poor water exchange. Net pen culture has been practiced worldwide in both marine and freshwater with little impact to substantial impact on water quality.

Benthic Environment

The best-studied aspect of marine net pen culture is the deposition of organic matter below the cages or pens. The culturing of animals in pens or cages results in some loss of feed and the production of feces. Uneaten feed and feces may accumulate under or near (usually no more than 30 m from the edge) the net pens or cages. Tidal flow, depth of site, composition, temperature, and salinity of the seawater are among the factors that influence the distribution of solids under each site. Accumulation of uneaten feed and feces can lead to anoxic sediments concurrent with changes in the microbial community and possibly leading to the release of methane, carbon dioxide, and hydrogen sulfide. As mentioned previously, there may or may not be links between water quality and the benthic environment. Therefore, those with more interest in this aspect are directed to Mahnken, who cited a number of studies covering nutrient dynamics and deposition with regard to the benthic community structure.

Methods of Reducing Impacts

As in other forms of aquaculture, methods to reduce the impact of aquaculture have been studied and proposed. Utilization of specifically formulated feeds and improving

feeding efficiency have been recommended to reduce the amount of waste produced in both the water and benthic environment. Another approach, which has been successful in Switzerland, is to diaper the cages or net pens. The most common approach to dealing with waste accumulation is the proper placement of farms. This is leading to more farms being sited in open water with better flushing rates and ability to naturally assimilate nutrients.

RECIRCULATING SYSTEM

Description

Recirculating systems are aquaculture systems in which the majority of water is treated or conditioned continuously for repeated use. One of the major advantages of recirculating systems is that a high density of fish biomass can be kept at optimum temperatures. The recirculating system is a relatively new technology that has been developed in the last two decades. There are various designs for recirculating aquaculture systems. An effective recirculating system will usually include a culture tank, an aeration or oxygenation system, a biofilter, and a suspended solids removal unit. The biofilter provides a medium for aerobic bacteria to break down organic fish waste and to convert excreted toxic ammonia into nitrate. Some systems will include additional treatment units such as disinfection.

Water Use

In a recirculating system, the water is processed continuously to guarantee optimum growing conditions. Water is pumped into the tanks through mechanical and biological filtration systems and disinfection systems and then returned into the tanks. Two terms that are often used in relation to recirculating systems are recirculation rate and water exchange rate. Recirculation rate (turnover time) is the amount of water per unit of time pumped out of the tank, through the biofilter, and back into the tank. This can easily be determined by dividing the volume of water in the tank by the capacity of the pump (in gallons per minute). An increase in recirculation rate will increase the number of turnovers per day in the culture tank and thereby provide reduced ammonia levels. Most production recirculation systems are designed to provide at least one complete turnover per hour (24 cycles/day). Exchange rate refers to the amount of water replaced per day. Most recirculating systems recommend 5–10% water exchange rate per day depending on the sensitivity of the species to water quality, stocking density, and feeding rates. The major consumption of water in recirculating systems is associated with the removal of solids such as backwashing the particulate filter or biofilters, while another major consumption derives from the need to replace evaporative loss.

Water Treatment

An excellent culture environment for fish is crucial. In the recirculating systems, the major unit

operations to consider for the maintenance of a good environment are ammonia removal/conversion and solids removal.

In respect to solids removal, waste solids, including uneaten feed and by-products of fish metabolism, have to be removed as quickly as possible. Left in the system, the solids could generate additional oxygen demand and produce carbon dioxide and ammonia nitrogen when undergoing bacterial decomposition. Solids removal is generally accomplished by the use of settlement, screen filters, or granular media filter such as bead filters or sand filters.

The most critical nonsolids water quality parameter is ammonia nitrogen. Ammonia is a principal excretory product of fish metabolism and in its un-ionized form is highly toxic to fish. Fish will excrete about 30 g of ammonia for each kilogram of food consumed (at 35% protein content in the feed). The most common technique used for ammonia removal is biofiltration that works on the principle of biological nitrification in which ammonia is oxidized successively to nitrite and finally to nitrate. Nitrate is much less toxic to fish, especially in freshwater. Another process called denitrification is an anaerobic process where nitrate is converted to nitrogen gas. Although not normally employed in commercial aquaculture facilities today, the denitrification process is becoming increasingly important as stocking densities increase and water exchange rates are reduced; resulting in excessive levels of nitrate in the culture system (29). Necessary conditions for nitrification are availability of dissolved oxygen, proper pH, and adequate alkalinity in the water. The rate of the biofiltration process is dependent on temperature, pH, salinity, surface area of the biofilter, and flow rate.

The types of biofilters commonly used in recirculating aquaculture systems include submerged biofilters, trickling biofilters, rotating biological contactors (RBCs), floating bead biofilters, and fluidized-bed biofilters. A common feature to all these biofilters is that they rely on a "fixed film" process in which nitrification bacteria develop on the surface of the support material of the filter and form an active bacterial biomass layer called biofilm. As fish culture water flows through the surface of the film, nutrients such as ammonia and oxygen will diffuse into the biofilm to be utilized by the nitrifying bacteria. The biofilm's importance also derives from the fact that, as a result of the bacterial attachment to a media, they are free from the risk of being washed out from the system when significant water exchange rate occurs.

Environmental Impact

Recirculation technology, predominantly used in land-based operations, enables farmers to achieve high efficiencies, providing them with greater control over the rearing environment. With the advent of recirculation technology for aquaculture, the incoming source water is reused to varying degrees and undergoes various treatment processes to ensure its purity. The farmer thereby gains considerable control over the rearing environment, being able to manipulate temperature, dissolved oxygen, photoperiod, and water clarity. With

more control over water parameters, the stocking densities can be increased, thereby allowing for greater economies of scale. Also, greater control is gained in the separation of uneaten feed and fecal material from the water, thereby preventing these nutrients from entering the downstream environment. Consequently, a recirculating system can significantly reduce effluent volume. However, a recirculating system cannot eliminate the production of fish waste. Thus, as with other animal production systems, a waste management plan is necessary for operation.

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BIOFUEL ALTERNATIVES TO FOSSIL FUELS

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INTRODUCTION

Our species has been exclusively dependent on biomass for fuel during most of its existence. The use of coal and mineral oil emerged due to the scarcity of biofuels (1). By now, the world energy market, which supplies a yearly energy consumption of about 400×10^{18} joule (J), is dominated by fossil fuels. Still, about 14% of yearly world fuel consumption is based on biofuels (2,3). Most of this consumption is similar to the use before the Industrial Revolution and is located in the countryside of developing countries. It is often associated with poor indoor and outdoor air quality. Negative impacts on soils, on the

availability of groundwater, and on water for generating hydroelectricity have also been linked to biofuels in developing countries (4–6).

Against this background, it may be somewhat surprising to see increased advocacy for biofuels. This concerns mainly “modern biomass”: biofuels applied while using advanced conversion technology to generate electricity and heat and for automotive power (7).

There are several main drivers behind the increased advocacy for biofuels. First, there are efforts to become less dependent on mineral oil, driven by concerns about energy scarcity, political instability in the Middle East, and oil price shocks. A second driver is environmental concern. Some applications of biofuels result in lower emissions of such pollutants as hydrocarbons, carbon monoxide (CO), sulfur oxides, and nitrogen oxides. It has also been argued that in combusting biofuels, one simply puts back into the atmosphere carbon dioxide (CO₂) that has been recently fixed by photosynthesis, thus ensuring climate neutrality, in contrast to the combustion of fossil fuels which puts CO₂ in the atmosphere that was fixed many millions of years ago. The need to find an outlet for agricultural surpluses has also been conducive to biofuel advocacy.

Advocacy and also actual combustion of “modern biomass” are increasing. A major application is the use of ethanol and to a lesser extent fatty acid methylester as fuel in motorcars. Ethanol is used mainly in Brazil, North America, and the European Union (EU). In Brazil, sugarcane is the major feedstock for ethanol production, and in the EU and North America, the main feedstock for ethanol is starch from grains. The use of biomass for modern electricity and heat production is mainly on the increase in the European Union. Expansion of biomass-based waste incineration, cocombustion of biomass in coal-fired power plants, and energetic use of residual biomass by industry contribute significantly to this trend.

Here the implications of modern biomass use that are relevant for water are discussed. Some of these implications are direct. By clear felling forests for the supply of biomass, there can be a major direct impact on surface water, such as changes in peak flows and runoff. Other implications are indirect. The extent to which modern biomass influences atmospheric levels of greenhouse gases such as CO₂ and CH₄ (methane) is a determinant of climate change and this in turn influences the biogeochemical water cycle. Current climate change leads among other things to sea level rise (that in turn may lead to increasing intrusion of saltwater) and an increase in land surface precipitation of 0.5–1%/decade in mid- and high latitudes of the Northern Hemisphere, whereas there is a decrease of such precipitation in subtropical land areas of 0.3%/decade (8). Due to climate change, there may be changes in water erosion, seasonal river flows, and groundwater tables; impacts on biological activity; and the presence of species in water. Increasing temperatures also increase human freshwater use (9).

In this article, we deal with the following questions:

- Is the use of biofuels “climate neutral”?
- What is the impact of biofuels on (fresh) water use?

- What is the impact of biofuels on substance flows that may impact water?

ARE BIOFUELS “CLIMATE NEUTRAL”?

To answer the question whether biofuels are “climate neutral,” a life cycle perspective on biofuel combustion is necessary. Such a perspective includes all stages of the biofuel life cycle. For instance, if trees from forests are used as biofuel, the cycle starts with the seed and ends with the postharvesting fate of the forest. The main matters to consider in establishing the impact of the biofuel life cycle on climate are the input of fossil fuels during the life cycle and the impact of the biofuel life cycle on biogenic carbon stocks. Both influence atmospheric concentrations of greenhouse gases such as CO₂ and CH₄. One should also consider the fate of compounds containing nitrogen (N) in the biofuel life cycle. Such compounds may be partly converted into the potent greenhouse gas N₂O (dinitrogen oxide). This aspect tends to be less important than the impact on atmospheric gases containing carbon, but may still be significant in agricultural production of biofuels.

Because the use of fossil fuels is so dominant in worldwide energy consumption, it is not surprising that usually there is an input of fossil fuels in the biofuel life cycle. Quantitatively, such inputs vary widely depending on the kind of biofuel and its actual production. For instance, in forest-derived biofuels, the input of fossil fuels is usually 5–10% of the energy value represented by the biofuel (output). This gives an “output to input energy ratio” of 10–20, meaning that the biofuel can supply 10–20 times the equivalent of the input of fossil fuels in the biofuel life cycle. In animal meal that is cocombusted with coal in a number of European power plants, the output to input energy ratio is about 0.24 (10). Table 1 gives output to input energy ratios for several biofuels.

This table shows, among other things, that for ethanol from U.S. corn, there is no agreement on the energy ratio. The underlying disagreements result in part from different estimates of the actual fossil fuel inputs in the biofuel life cycle. When corn is used to produce the biofuel ethanol, there are also coproducts, such as cattle feed. In this case, the allocation of fossil fuel input to the different outputs (products) arises. One possibility is to allocate all fossil inputs to the biofuel. Another possibility is to divide the fossil fuel inputs by the products involved. Such an allocation may be done in different ways, for instance, based on a monetary or on an energy basis, and these different ways lead to different outcomes. For ethanol produced from corn in the United States, the U.S. Department of Agriculture (12) has argued that the biofuel energy output to fossil fuel energy input ratio is currently 1.3, whereas Pimentel (13) calculates an output to input energy ratio of 0.58–0.74, depending on the allocation to ethanol and its coproducts.

However, notwithstanding this disagreement, one may conclude that when ethanol is produced from U.S. corn starch, the input of fossil fuels is at least a large share of the energy output from the biofuel. A similar conclusion can, for instance, be drawn regarding oil-methylesters

Table 1. Ratios of Current Energy Output to Fossil Fuel Input for the Life Cycles of a Number of Biofuels^a

Type of Biofuel and Application	Ratio of Energy Output to Fossil Fuel Input
Forest derived (short transport distances), for electricity production	10–20
Ethanol from Brazilian sugarcane (bagasse as production fuel), transportation fuel	~7
Ethanol enzymatically produced from woody crops, transportation fuel	~4
Chicken manure (energetic allocation) produced in the European Union, for electricity production	~1
Ethanol from U.S. corn, transportation fuel	0.58–1.3
Animal meal produced in the European Union for electricity production	0.24

^aReferences 3, 10–13.

produced from oil crops such as rapeseed, sunflower, or soybean (14). More in general, one can conclude that burning biofuels is not only a matter of giving back to the atmosphere carbon that was recently fixed. Usually, there is the added emission of fossil fuel derived carbon compounds, such as CO₂. Especially when high-input agricultural production is involved and when further processing is fueled by fossil fuels, this emission tends to be important.

A second important matter to be considered is the effect of the biofuel life cycle on biogenic carbon stocks: biomass and soil organic carbon (C). The last stock is of great quantitative importance. It is currently about three times the atmospheric stock and about four times the stock of C in biomass. Since the beginning of the Industrial Revolution, the combustion of fossil fuels has added an estimated 270×10^{15} gC to the atmosphere, whereas biogenic sources (deforestation, biomass burning, and loss of soil organic carbon) added $\sim 136 \times 10^{15}$ gC. Depletion of the stock of soil organic C since the start of the Industrial Revolution is an estimated 78×10^{15} gC (15).

The biofuel life cycle can substantially reduce and add to biogenic carbon stocks. If biofuel is produced from deforestation and conversion into agricultural land, large emissions of carbon containing gases from soil to the atmosphere can occur. On the other hand, when biofuel is produced from reforestation or conservation tillage and the use of compost and manure, there can be net sequestration of carbon in the soil (15). In Table 2, this is exemplified by the life cycle emission of 68×10^2 g CO₂/kWh, when the production of forest derived fuel is accompanied by deforestation. The net sequestration of 19×10^2 g CO₂/kWh is from reforestation of agricultural land.

If we survey the relation between biofuel use and the emission of greenhouse gases that affect climate, it will be clear that the relation is more complicated than “giving back” recently fixed carbon dioxide to the atmosphere. As Tables 1 and 2 show, net life cycle additions of greenhouse gases to the atmosphere can be

Table 2. Life Cycle CO₂ Emission Factors for Several Current Types of Power Generation in 10² g CO₂/kWh^a

Type of Fuel	Life Cycle Emission Factor in 10 ² g CO ₂ /kWh
Hard coal	+10
Natural gas	+ 4
Forest derived biofuel	–19–+68

^aReference 3.

larger than that from combusting an equivalent amount of fossil fuel.

THE IMPACT OF BIOFUELS ON (FRESH) WATER USE AND WATER AVAILABILITY

The replacement of forests by plantations or annual biofuel crop production may lead to a decrease in available water due to increased drainage and increases in peak flows and storm flows. Soil degradation from clear felling of forests may lead to insufficient replenishment of groundwater reserves and increased peak and storm flows (16,17).

Water is also an important resource in biofuel production. Fast growing woody crops such as poplar and eucalyptus use relatively much water. For instance, Tuskan (18) has calculated that in the United States hybrid poplar requires roughly 1 meter/hectare yearly of (ground) water during midrotation. In southern China, it has been found that under comparable conditions, water tables under eucalyptus plantations were 50 cm lower than under mixed forests (19). Water use for biofuel crops may be expected to increase when temperatures rise (20).

Although there are some waterlogged areas where large uptakes of water by energy crops can be considered a benefit, it will be more common that water can be a limiting factor in biofuel production. Currently, in nearly 80 countries that have 40% of the world population, water demands already exceed additions to stock (21), and such countries are unlikely to embark on major energy crop production programs. In countries that currently have surplus water, if compared with demand, limitations may also emerge. For instance, in Ethiopia where eucalyptus is grown for biofuel production, competition between food crops and eucalyptus for scarce water resources has led to restrictions on the latter (4). Berndes (22) has pointed out that large-scale biofuel production countries such as Poland and South Africa may face absolute water scarcity, resulting in lowering of water tables.

Judicious choice of biofuel crop may limit water use. C₄ biofuel crops generally use water more efficiently than C₃ crops (22). Water efficiency may also be improved by some types of agroforestry, reducing runoff, direct evaporation of water, and drainage (17,23).

BIOFUEL SUBSTANCE FLOWS THAT MAY IMPACT WATER

The biofuel life cycle mobilizes large amounts of substances. When biofuels are produced by agriculture (e.g., ethanol made from starch or sugar), higher inputs of

sediments, nutrients, and pesticides in water accompany increased agricultural land use. Such increases may also be large because land requirements may be large. If, for instance, the United States were to decide to cover current U.S. gasoline consumption by corn-based ethanol fuels, more land would be needed than current total U.S. cropland (11).

When biofuels are derived from tree plantations, erosion may be much increased. Studies of plantation forestry on slopes has shown that, especially in the planting and timber harvesting phase, erosion can be extensive, giving rise to silting of lakes and reservoirs and channel instability downstream (24). The sequestration of minerals may be reduced by clear felling, leading to increased levels in surface water (25).

Another matter for concern in tree plantations is the use of plant nutrients to stimulate biomass production. High productivity is strongly linked to large inputs of plant nutrients containing K (potassium), N, and P (phosphorus), and to poor efficiencies in output (nutrients in biofuel) to input ratios (16,26). This may lead to major nutrient losses to water.

The actual impact of biomass production, of course, strongly depends on the measures taken to limit such impacts. Woodlands generating yearly, on average, 3 metric tons of woody biomass per hectare will lose much less nutrients and soil than high productivity plantations (27). In the agricultural production of biofuels, conservation tillage, applying crop residues, terracing, and other provisions to trap sediment, reduce fertilizer application, and minimize hydrologic changes may limit impact.

The combined life cycle emissions of acidifying substances such as NO_x and SO₂ are for biofuels of the same order of magnitude as the corresponding emissions from fossil fuel life cycles (28–30). The generation of persistent organic pollutants such as chlorinated dioxins and benzofurans and polycyclic aromatic hydrocarbons is not less than that from fossil fuels (31,32). The mobilization of heavy metals and other hazardous elements may be substantial. In Denmark, for instance, the application of fly ash originating in the combustion of wood and straw to soil is banned because of its cadmium content (33). Klee and Graedel (34) have compiled a survey of elements in coal and the average concentration in dry plant material. Significant differences emerge from this survey. For a number of hazardous substances, including Hg, As, Pb, Cr, and Ni, concentrations in biomass are lower. On the other hand, there are also elements that may have an environmental impact that have higher concentrations in biomass, such as P, K, Co, and Cd.

The load of hazardous elements generated by biomass combustion may be much increased when such elements are added to biomass during its stay in the economy. Examples are the treatments of wood with chromated copper and arsenic salts or metaborates for conservation. When wastes of such treated woods are burned as biomass especially the level of hazardous elements in ash is elevated. Sewage sludge that is combusted as biomass is another case in point.

Again actual impacts will depend on the measures taken. For instance, state-of-the-art use of desulfurization

and de-NO_x technology will strongly limit acidification risk for poorly buffered surface waters. And indefinite containment of hazardous elements in ashes will better protect ground- and surface water than application of ashes to soils or the use of settling ponds for ashes.

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SOIL CONSERVATION

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INTRODUCTION

Soil is a precious natural resource, which performs many vital functions. As a growing medium, soil provides nutrients, air, and water to sustain life on earth, supporting

global agricultural and manufacturing industries with renewable raw materials, enabling sustainable economic growth and prosperity. As a habitat, soil is an essential gene reservoir, maintaining a healthy biodiversity that enables micro- and macroorganisms to adapt successfully to environmental changes. Soil influences our natural and cultural landscapes by providing the foundation and raw materials for infrastructure, thus affecting the aesthetics of the world around us. Soils have enormous storage capacity, absorbing surface water, which helps to reduce flooding risk, and providing water supplies for agricultural, domestic, and industrial demands. Soil filters the water passing through it, and chemical transformations help neutralize potentially harmful substances. Soils also store nutrients, trace elements, and organic carbon. Soils are essential in the sequestration of carbon and in the absorption, storage, and reflection of solar energy—important processes that influence global climate and climate change.

Soil formation rates are very low, roughly equivalent to $0.1 \text{ mm} \cdot \text{y}^{-1}$ in temperate regions (1). With such limited rates of renewal, soil should be treated as a nonrenewable resource, which must be protected and conserved so that the functions of the soil, as listed above, can be maintained for future generations.

THE NEED FOR SOIL CONSERVATION

Soil conservation aims to prevent degradation of the soil resource. Such degradation occurs when one or more of the functions described above are impaired, either through *in situ* changes in soil properties, such as loss of soil organic carbon or depletion of nutrients, or through physical loss of the soil through erosion. As the latter process is irreversible, emphasis on erosion control is given here. Erosion is initiated when the kinetic energy from running water or wind is sufficient to detach and transport soil particles and aggregates from the soil mass. The eroded soil is transported and some or all of it may be deposited downslope. Often, however, the finer particles, mainly clays and organic matter, are transported into adjacent watercourses, where the nutrients associated with them can cause pollution, and deposition of the soil particles (sedimentation) can restrict water flow.

Soil conservation aims at conserving the soil properties associated with fertility (Table 1) by preventing *in situ* degradation and preventing physical loss of soil by controlling erosion. Ideally, soil conservation should also improve the levels of these fertility indicators to ensure continuing soil health and functionality and to minimize the detrimental impact of any future soil degradation process.

SOIL CONSERVATION APPROACHES

Soil conservation techniques can be classified into three approaches:

1. *Mechanical Methods*—the use of engineering structures to conserve fertility and control soil erosion.

Table 1. Main Components of Soil Fertility

Physical components	Soil depth
	Bulk density
	Soil water/water holding capacity
	Aggregation/soil structure
	Porosity/soil air
Biological components	Microorganisms
	Macroinvertebrates
	Soil respiration
	Organic matter/carbon
Chemical components	Extractable nutrients
	Essential elements
	Cation exchange capacity
	pH
	Electrical conductivity
	C/N ratio

2. *Agronomic Methods*—the use of vegetation or simulated vegetation to conserve fertility and control soil erosion.

3. *Soil Management Methods*—the manipulation of soil conditions to conserve fertility and control soil erosion.

These methods are interrelated and multipurpose in that they aim to conserve both soil fertility and control soil erosion simultaneously. In many situations, a single approach is insufficient on its own and several approaches need to be combined in an integrated soil conservation system.

Mechanical Methods of Soil Conservation

Introduction. Mechanical methods of soil conservation are applied to control soil erosion by water. Some techniques date back to ancient times such as terraces, which were adopted by Phoenician, Roman, Chinese, and Incan civilizations but are seldom used in modern times, due to high capital and labor costs incurred in construction and maintenance.

Types of Mechanical Soil Conservation Measures. Terraces are constructed on hill slopes where soil conservation is needed because of erosion risk. Terraces control soil erosion by reducing slope length and local slope steepness, thus reducing overland flow energy to erode and maintaining inherent soil fertility.

There are many different types of soil conservation terrace, often developed for local environmental conditions (e.g., the “fanya juu” terrace of eastern Africa). Generic types of terrace include the channel and bench terrace. Channel terraces are suited to slopes no steeper than 7° (12%) (2,3) (Fig. 1). They consist of a cut channel and a filled ridge (constructed from the cut material), excavated across the slope.

The distance between consecutive terraces downslope can be determined by equations based on site conditions (4,5). However, a general rule of thumb is to use a vertical interval between terraces of 1 m. This results in varying distances between terrace channels as the slope gradient changes, with shorter terrace intervals on

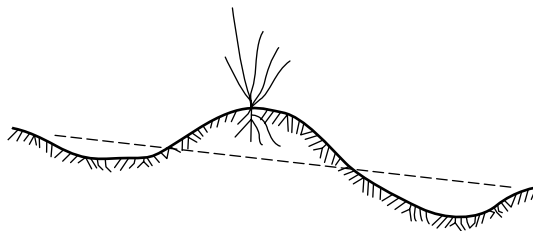


Figure 1. Cross section of a channel terrace (<http://www.fao.org/documents>).

steeper slopes. This may cause practical problems for mechanized farming.

The terrace channel reduces slope length, and overland flow generated between terrace channels discharges into the channel. The size (width × depth) and gradient of the channel across slope (usually at 1: 250 = 0.4% slope) determine its capacity in conveying overland flow to a safe outlet (see lined waterways below). The gentle gradient of the channel ensures that erosion within the channel itself does not occur.

Bench terraces are steplike structures, constructed across slope, which prevent soil erosion by dissecting the slope into smaller, shorter components, each with a level or near-level bench (Fig. 2). The dimensions of the benches depend on slope steepness, the desirable width of cultivable land (i.e., bench width), and the acceptable height of the steep wall of the terrace, or “riser” (4,5). The construction of the level bench requires the oversteepening of the terrace riser. To prevent the collapse of the riser, it must be supported with stones, rocks, masonry, timber, or simply a dense network of vegetation roots.

By concentrating and redirecting overland flow, terraces will change the natural hydrology, which may incur increased risk of erosion. To avoid this, flow in the terrace channels is directed to lined waterways, which are designed to carry increased volume and velocity of overland flow without causing erosion. These waterways, ideally located in natural depressions to minimize earth moving, are lined with dense, short vegetation, stones, or geotextiles. These linings permit high velocity flows without causing erosion. The width and depth of the waterway are determined by the volume of overland flow and the waterway slope and lining.

In conclusion, mechanical measures of soil conservation control the physical loss of soil by erosion. As the soil is kept *in situ*, soil fertility is also maintained.

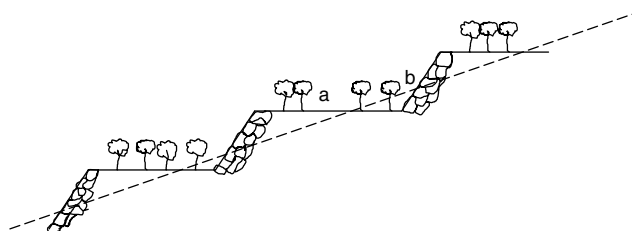


Figure 2. Cross section of bench terraces: a—level bench; b—riser.

Agronomic Methods of Soil Conservation

Introduction. Vegetation is highly effective at controlling soil erosion (6). Agronomic methods of soil conservation involve the use of vegetation to control soil erosion and to maintain or enhance soil fertility.

Water Erosion Control. The canopy, stem, and root components of plants interact with soil erosion processes. A plant canopy intercepts rainfall, changing raindrop sizes and reducing fall velocities, so that rainfall kinetic energy available for erosion is reduced. Plant stems distort overland flow paths and impart roughness to the flow, reducing flow velocity and thus kinetic energy to erode. Plant roots encourage infiltration, reducing overland flow generation, and also increase soil shear resistance against the shearing forces of overland flow. These beneficial effects of vegetation have been quantified by many workers (7–10). A fully mature, low growing, dense vegetation cover is able to control soil loss by two or three orders of magnitude compared to bare soil.

Agronomic methods of soil conservation apply these principles in the field. Maximum cover on the soil surface will protect the soil from erosive raindrops and overland flow, absorbing their energy and thus reducing the ability to erode. High stem and root densities reduce overland flow volumes, velocities, and energy to erode. These effects are achieved through cover cropping, high density planting, crop rotations, mulching, and erosion control mats. All of these control soil erosion and enhance soil fertility by contributing nutrients, organic matter, and carbon to the soil, thus encouraging microbial biomass and activity and improving soil structure and aggregate stability. A fertile, noneroding soil will be able to support a dense vegetative sward with high percentage cover, stem, and root densities, which in turn are effective at controlling erosion and enhancing soil fertility.

Vegetation is used to reduce slope length and steepness when grown in strips parallel to the contour. The strips may be of different crops (strip cropping) or as a single field strip across the slope. The strips restrict the generation of overland flow and encourage deposition of eroded sediment, so that local slope gradients might be reduced. The deposited sediments are also rich in nutrients, which are retained within the field, thus maintaining fertility and avoiding diffuse pollution off-site.

Agroforestry systems combine erosion control and maintenance of soil fertility (11). One common practice is to grow perennial trees/shrubs alongside annual crops. The trees form across-slope barriers to overland flow (also known as “alley cropping”), thus reducing erosion. While the annual crops may deplete soil nutrients in the topsoil, these can be replaced by release of nutrients assimilated in the leaves that fall from the trees. These nutrients come from reserves deeper in the soil by the tree roots, so competition for nutrients between the two components is minimal.

Wind Erosion Control. By exerting a drag on air flow, vegetation can reduce the velocity of wind below that required to initiate erosion and can enhance the deposition of soil already in the air. The most common

use of vegetation is in the form of shelterbelts spaced at regular intervals at right angles to the direction of the erosive winds. Traditionally, trees have been used to form shelterbelts but they need to be supplemented by shrubs (bushes) and grasses to provide cover close to the ground surface, as 70–75% of the soil eroded by wind is carried within the lower 1 m of the atmosphere. Modern shelterbelts are based on species of willow and alder, both of which can be coppiced to encourage growth close to the ground. The most important characteristic of a shelterbelt is its density or porosity. The belt needs to be porous so that some air passes through the belt but at a low velocity. Clearly, if the belt is too sparse, the velocity of the wind is not reduced. A porosity of 40–50% is generally considered optimum. Belts are usually designed to maintain wind speeds below 75% of the speeds recorded over open ground. To achieve this, they need to be spaced at intervals of between 12 and 17 times the belt height.

The principles of shelterbelts can also be used to design in-field shelter systems in which crops are alternated on a strip system, aligned at right angles to the prevailing wind. Thus, strips of barley, spaced at 5–10 m intervals, can be alternated with strips of carrots, onions, or sugar beet to control erosion.

Since shelterbelts require a certain amount of time to become effective, they are not appropriate where immediate control of wind erosion is required. In these situations, windbreaks can be made from brushwood or plastic netting. Brushwood fences are commonly constructed on coastal dunes to reduce wind velocities and stabilize the mobile sand, which can then be seeded or planted with shrubs and trees to provide long-term stability and reclamation.

Soil Management Methods of Soil Conservation

Farming operations carried out up and down the slope will increase the risk of overland flow generation and loss of soil and nutrients. Changing to contour cultivations will reduce these risks significantly, but this is not always practical, especially on complex or steep slopes, where the arrangement of fields on the contour makes harvesting difficult and incurs high energy consumption. Contour ridging creates raised ridges across slope, which will reduce slope length and an effective catchment area for overland flow generation, and intercept any overland flow that is generated. The across-slope ridges can be joined together at regular intervals by “ties” running up/downslope, thus creating small basins that trap any overland flow and eroded soil, conserving both soil and water within the “tied ridges.”

The factors affecting the susceptibility of soil to erosion can be affected by soil management methods, which can both degrade or conserve soil. For example, intensive arable production can lead to severe soil degradation, as soil is broken down by numerous tillage practices into a loose, fine tilth seedbed. The resulting small soil aggregates are highly susceptible to erosion. Studdard (12) and Pochard (13) showed that under conventional arable production, there is little time between cultivations for aggregates to reform into larger, more stable, less erodible units. Soil structural stability is further reduced by loss of

carbon by oxidation as tillage exposes deeper soil layers to the air. Soil faunal biomass, including fungal mycelia and earthworms, is important in maintaining cohesive bonds within and between aggregates, but it can be destroyed during soil disturbance.

Reducing the number and/or intensity of cultivations preserves soil structural stability and aggregation. Less disturbed aggregates are able to resist erosive forces, as cohesive bonds within the aggregates are maintained. Fewer, less invasive cultivations mean more of the vegetative biomass remains on or within the soil. This biomass is able to absorb the energy of erosive agents, so erosion is controlled. For example, a rough, trashy soil surface imparts friction to overland or air flow, causing turbulence and eddying, resulting in a reduced flow velocity and kinetic energy to erode.

The relative reduction in soil disturbance and retention of vegetative material on and within the soil are the principles behind the use of "conservation tillage." This term incorporates a continuum of soil management methods such as minimum till, zero till, mulch tillage, direct drilling, noninversion tillage, subsurface mulch tillage, and plough/plant cultivation (14). Conservation tillage aims at maintaining soil fertility and controlling erosion. It can also reduce input costs to agriculture, as fewer, less invasive cultivations expend less energy and fuel, resulting in dramatic adoption rates in the 1970s in the United States during the oil crisis. It should be noted that the success of conservation tillage is related to soil type and timeliness of operations. For example, there is evidence that reducing the number of cultivations can increase bulk density on structureless soils, leading to overland flow generation and flooding risk, which can only be alleviated by deep subsoiling periodically. Supporters of conservation tillage claim that these are short-term effects, which are offset in the medium to long term by the improvement of aggregation and inherent soil structure resulting from increased levels of biotic activity and organic matter levels.

Manipulation of the soil surface condition by tillage is used to control wind erosion. On light soils such as sands and silts, the soil is ploughed and then pressed with a heavy roller. Drilling is then carried out at 90° to the plough/press operation. The resultant tilth is rough and cloddy, to impart maximum friction to air flow, thus reducing wind speed and ability to detach and transport soil.

Soil conditioners are applied to soils susceptible to degradation, especially erosion. These products can perform the dual role of enhancing soil fertility and controlling erosion. Products with high organic matter (such as seaweed extract and tree bark) improve inherent soil fertility in terms of nutrients and organic matter, thus improving aggregation, structure, porosity, infiltration capacity, and water holding capacity.

Hydrophilic soil conditioners including bark, limestone, palm oil effluent, and seaweed extract increase aggregation and infiltration capacity, so overland flow is less likely to occur. The increase in organic matter from these products helps control erosion by increasing cohesive bonds

Table 2. Objectives of Soil Conservation Measures in Relation to Economic and Social Factors

1. The need for soil conservation should be recognized by all stakeholders—farmers, local communities, landowners, advisory services, government departments, NGOs, and politicians.
2. Soil conservation measures must be compatible with present and future management plans: they should not be imposed by external bodies such as researchers, scientists, advisory and extension services, or politicians.
3. Methods of construction will be based on available and appropriate technology and level of education. Technology transfer will be "bottom-up" not "top-down."
4. The easiest and minimal maintenance of soil conservation scheme will be adopted.
5. Present infrastructure should not be radically changed by soil conservation measures.
6. Mechanical measures should be laid out economically.
7. Crop selection will be based on economic as well as soil conserving properties.
8. Available grants and subsidies will be used. This has been demonstrated in the United States for many years. Farmers are eligible for USDA farm subsidies, but only if they demonstrate soil conservation practices have been implemented. A similar scheme will operate in England and Wales under the DEFRA Single Payment Scheme. To be compliant with the scheme, farmers have to draw up a simple risk-based soil management plan that protects against soil erosion and maintains soil organic matter and soil structure.
9. Conservation measures will ensure long-term productivity. Farmers are considered "land stewards," taking responsibility for conserving the soil resource for future generations. This concept is exemplified by the "Landcare" movement in Australia and New Zealand.

and the stability of aggregates. Hydrophobic soil conditioners including latex, emulsions, and molasses form an impermeable film on the soil surface, which repels water and increases the soil's resistance to erosion. While these latter products control soil erosion, they have no effect on inherent soil fertility.

IMPLEMENTATION AND ADOPTION OF SOIL CONSERVATION MEASURES

While much is known of soil conservation techniques, they will fail unless the right social, economic, and political conditions prevail (15). Table 2 highlights the main considerations for the adoption and long-term success of soil conservation measures.

CONCLUSIONS

Soil is a sustainable resource, if treated carefully and with respect. Failing to protect and conserve soil can lead to rapid soil degradation, which in turn can be an irreversible process. Soil conservation can be defined as the maintenance of soil fertility and the control of soil erosion. These two objectives can be realized by appropriate management of the soil resource, using diverse soil conservation approaches. Application of these techniques

will ensure that the diverse forms and functions of soil will be conserved for future generations.

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LANDSCAPE WATER-CONSERVATION TECHNIQUES

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When given information and technology, people will make the effort to conserve. Plant selection will have less impact on water use than either irrigation or soil preparation. Water budgets give consumers more options and personal choice, while providing an effective way to conserve water.

Outdoor water-conservation measures typically focus on reducing or eliminating landscape water use. But implementing new scientific findings and advanced

technology and general education can go a long way toward achieving the same end, just as these methods have proved successful in conserving water indoors. People will act to conserve water and improve the environment when properly informed of and motivated by the best scientific knowledge and technology.

AN INDIVIDUAL'S RIGHT TO CHOICE

Numerous water-use studies have documented that, depending on an area's climate, residential outdoor water use can account for between 22 to 67% of total annual water use. Clearly, this represents a vast opportunity for conservation. But to maintain an individual's right to personal choice and to maximize the positive environmental benefits of landscaping, a variety of factors need to be addressed that take location into account. When dealing with living plants, a one-size-fits-all solution will not be effective. But proven advanced scientific landscape water-conservation principles and practices do exist, and these can be modified and refined according to area-specific needs (Fig. 1).

THE NEED FOR CLEAR AND CAREFUL DEFINITIONS

First, however, potential targets for widespread outdoor water conservation should be clearly and carefully defined. Too often, a narrow definition focuses exclusively on landscape water use. Narrow definitions often overlook potentially high water-use elements such as swimming pools or other water features, whose evaporative losses are as great as or greater than those from landscape application.

Other nonlandscape outdoor water uses include washing cars, driveways, sidewalks and siding—and even some types of children's water toys. In addition to conserving in these areas, many techniques can be applied landscape water conservation as well.

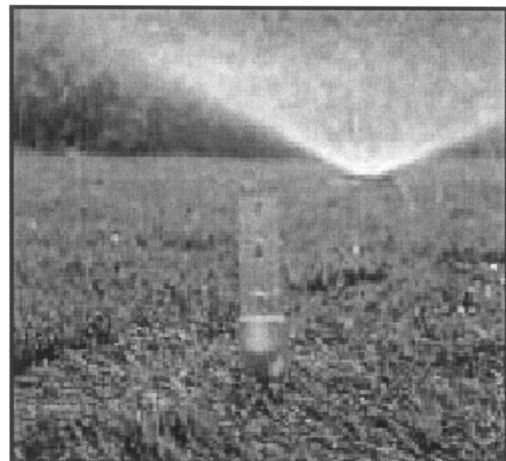


Figure 1. Rather than attempting to regulate or ban a specific water use, the water budget technique leaves the determination in the hands of the rate-paying water user.

THE WATER BUDGET PROGRAM

One of the most effective techniques is based on the practice of advising people how much water they can use, rather than telling them how they must use it. Termed a “water budget” or “water allocation” method, water providers establish a series of escalating allocation/pricing tiers so that every unit of water (i.e., 1000 gallons) in excess of a base quantity costs more than the previous unit.

Because outdoor water use can be measured and priced higher, people adjust their end uses according to their personal desires and financial concerns. This approach eliminates the need for contentious public hearings on landscape ordinances and the development of debatable plant lists, as well as the potential for draconian enforcement practices and so-called “water police.”

Water budgets, for both indoor and outdoor water use, encourage individual freedom of choice and allow artistic expression by homeowners and landscape designers. Rather than attempting to regulate or ban a specific water use, this technique leaves the determination in the hands of the rate-paying water user. As we’ve seen, when gasoline prices rise, individuals can quickly adjust their use patterns. The same holds true for water.

Once purveyors decide how to allocate and price water, they then have the very important role of assisting in the development and distribution of scientifically based educational materials on water conservation. Again, successful indoor conservation practices can be easily converted to outdoor water conservation, particularly as it relates to landscape water use.

The Two-Track Strategy

Moving from general to more specific landscape water-conservation recommendations, a two-track strategy that emphasizes different approaches for existing landscapes and newly planned and installed landscapes may be most effective, as follows.

Existing Landscape Areas

Predrought/Premaximum Heat-Day Practices

1. **Increase water infiltration** with dethatching or hollow-core aerification of all lawn areas, as well as under the drip line of trees. Till garden areas to break up surface crusting, adding mulch where appropriate.
2. **Trim or prune** trees, shrubs, and bushes to remove low-hanging, broken or diseased parts and allow greater sunlight penetration throughout and beneath the plant. Generally speaking, plant water use is proportional to total leaf surface; thus, properly pruned plants should require less water.
3. **Fertilize all plants** (when soil temperatures reach at least 50 °F or 10 °C) with a balanced plant food that contains nitrogen (N), phosphorus (P), and potassium (K) according to the results of soil testing or as experience has shown is appropriate.
4. **Sharpen pruning shears and mower blades.** Dull blades encourage plant water losses and the introduction of disease.

5. **Establish or confirm soil type(s)** to match water-infiltration rates with future water-application rates and to determine if soil pH adjustments are recommended.
6. **Perform irrigation-system maintenance**, regardless of type (hose-end, drip, in-ground, etc.) to ensure maximum uniformity of coverage and overall operation. Repair or replace broken or damaged nozzles or heads. Flush drip system emitters to ensure proper flow. Ensure that rainfall shutoffs and other devices are working properly.
 - a. Acquire and/or install hose-end water timers for all hose bibs.
 - b. Adjust in-ground system controllers according to plant’s seasonal needs.
7. **Upgrade in-ground irrigation systems** by adding soil-moisture meters, rain shutoff devices, or evapotranspiration (ET)-based controllers.
8. **Relocate drip emitters**, particularly around trees, to the outer edge of their drip lines. This will result in higher water use, but it will also encourage a better root system that will anchor the tree in high winds and provide the water and nutrients that are needed by large trees, shrubs, and other plants (Fig. 2).



Figure 2. Locate drip emitters to provide water and nutrients needed by large trees, shrubs, and other plants.



Figure 3. Water in the early morning.

9. **Confirm water-application rates** for hose-end or automatic systems to know the actual running times required to distribute a specified amount of water within a given amount of time.
10. **Water in early morning** (Fig. 3) when wind and heat are lowest, to maximize the availability of the water to the selected plants.
11. **Irrigate all plants infrequently and deeply**, according to local ET or soil-moisture requirements, to establish a deep, healthy root system. A core-extracting soil probe or even a simple screwdriver can help determine when to water if the more sophisticated ET rates are not available (Fig. 4). Professional turf managers gradually lengthen the interval between irrigations to create gradual water stress for deeper rooting.
12. **Cycle irrigation applications** (on-off-on-off) to allow penetration and avoid runoff. Depending on soil types, the running times may be from 5 to 15 minutes and off times from 1 to 3 hours. Repeat this cycle until necessary amounts of water are applied and maximum penetration is achieved.
13. **Begin regular mowing** when grass blades are one-third higher than desired postmowing length, and keep clippings on the lawn.
14. **Raise the mowing height** as the summer progresses to the highest acceptable level to encourage deep rooting. (Note: This has been a traditional recommendation, but further study is required to refine this approach and maximize effective water use and/or conservation.)

Drought or Maximum Heat-Day Practices—to Maximize Landscape Appearance

1. **Withhold fertilizers**, particularly nitrogen, on turfgrass; however, small amounts of potassium will aid in developing more efficient roots.
2. **Reduce mowing frequency** to minimize shock to turf areas.
3. **Reduce traffic on turf areas**, this will minimize wear and possible soil compaction.
4. **Adjust automatic timers** of in-ground irrigation systems according to plant's seasonally changing water needs.

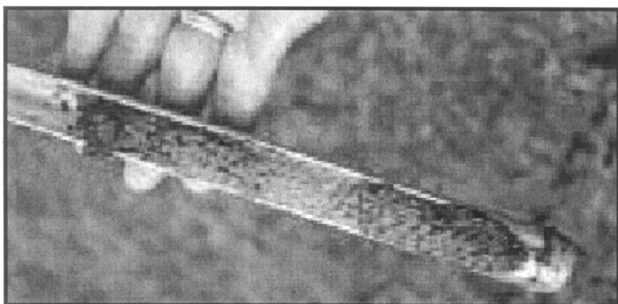


Figure 4. An example of a core-extracting soil probe.

Drought or Maximum Heat-Day Practices—If Dormant Turf Appearance is Acceptable

1. **Eliminate all traffic on turf areas**, including mowing, which will probably not be necessary because of the extremely slow growth rate.
2. **Adjust automatic timers to manual** or use hose-end sprinklers to apply approximately one-quarter inch of water a week. The dormant lawn will have a tan, golden, or light brown appearance; however, light/infrequent watering will be sufficient to maintain life in the crowns of the grass plant during this period.
3. **Minimize water applications** for all plant materials to the essential amounts needed to maintain plant vitality.

Postdrought or Maximum Heat-Day Practices Irrigate All Plants to Reestablish Soil-Moisture Levels, beginning with staged increases in watering and progressing toward deep and infrequent watering. By gradually lengthening the running times but adding greater spacing between watering applications, the initially shallower roots will extend to reach deeper soil moisture. Previously dormant turf will recover rather quickly, and other plants will regain their vigor.

Newly Planned or Installed Landscape Areas. Though fewer in number than existing landscape areas, newly planned and installed landscape areas can generally achieve greater water savings if all of the best design, plant selection, installation, and management practices now available are closely observed and fully implemented.

The basic principles of xeriscape landscaping provide an excellent starting point, providing they are fully understood and properly applied. However, note that it is incorrect to assume that a xeriscape is supposed to be a totally grassless landscape or one that uses only rocks, cactus, or driftwood.

Dr. Doug Welsh, former director of the National Xeriscape Council, wrote in *Xeriscape Gardening: Water Conservation for the American Landscape*: "In xeriscape landscaping we try to plan the amount of turf so that the investment in water will be repaid in use and beauty. In many instances grass is the best choice. For play areas, playing fields and areas for small pets, grass is often the only ground cover that will stand up to the wear. Turf also provides unity and simplicity when used as a design component."

Rather than duplicate information contained in several xeriscape manuals, this publication will focus on a limited number of very critical elements that can maximize landscape water conservation.

Efficient irrigation is without question an important water-conservation activity. People waste water; plants don't. Overwatering not only wastes water, but it also weakens or kills plants more than underwatering. Another wasteful practice seen all too often is misapplication of water, resulting in rotted fences and house siding, flooded



Figure 5. A wasteful practice seen all too often is misapplication of water, resulting in flooded sidewalks or driveways and rivers of water wastefully flowing down gutters.

sidewalks or driveways and rivers of water wastefully flowing down gutters (Fig. 5).

Though less so today, many new in-ground landscape-irrigation systems have been sold in the recent past on the basis of simplicity, for example, “set it and forget it.” Homeowners, intimidated by the sophisticated appearance of the system’s control box, would not modify the settings for seasonal plant water-use changes. Even worse, to be cost-competitive, many systems did not include readily available and relatively inexpensive soil-moisture meters, rain shutoff devices, or multistation programs. These deficiencies have resulted in overwatered landscapes, where water runs down the streets and systems continue to operate during torrential downpours.

Because of its high visibility, turf watering can be seen as the antithesis of water conservation and is often an out-and-out target for elimination. In some locales, “cash-for-grass” programs are used to pay homeowners handsomely to remove grass from their landscapes. One highly respected West Coast water official noted at a conservation convention: “It isn’t the grass that causes a problem, it’s the poorly designed and poorly operated irrigation system. I can’t control the irrigation systems, but I can reduce the amount of grass in a landscape, and that will control the water-use problem created by bad irrigation.” The fault was not the grass, but the fact that it was being improperly watered.

Soil analysis and improvements are another very important aspect of water conservation. The soil on most new residential and commercial landscape sites has literally been turned upside-down during the construction process, and the topsoil is placed beneath a layer of clay. Then it is compacted as hard as cement by equipment, piles of building materials, and construction-worker foot traffic. The soil’s texture, chemistry, and natural flora and fauna are destroyed.

More water could be conserved and healthy landscape plants more easily grown by improving the soil before planting than by any other process or technique. The initial cost of adding topsoil and soil test-guided amendments to

improve the soil may seem high, but the return on that investment will be even higher. Failing to improve the soil prior to planting, when it is most practical and efficient, will result in roots not being able to penetrate as deeply as possible and runoff occurring almost instantaneously. If the soil pH is not correct, plants will not be healthy, nutrients will not be used, and chemical leaching can take place.

Appropriate plant selection can be a source of frustration or misunderstanding and not produce the hoped-for water savings, particularly when it is combined with “practical turf areas.” Lists of low water-use landscape plants can cause confusion because they are based on the incorrect assumption that those plants capable of surviving in arid regions are low water users, when the plants typically are only drought-resistant (Fig. 6).

The majority of turfgrass species and cultivars have been scientifically assessed for their evapotranspiration (water-use) rates and can be selected according to the needs of a specific climate. On the other hand, very few tree and shrub species and cultivars have undergone comparable quantitative water-use assessments. One stunning exception comes from research at the University of Nevada, where Dr. Dale Devitt found that “one oak tree will require the same amount of irrigation as 1800 square feet of low-nitrogen fertilized turfgrass!”

It is also important to understand that though “low water-use” plant lists were developed with the best intentions and purposes in mind, practically all of these lists have been based on anecdotal evidence or consensus judgments, not scientific measurement under controlled procedures. Quite simply, a given group decides that based on their experiences and suppositions, a certain



Figure 6. The minilysimeter gauges water use under actual turf conditions. This photo at Washington State University’s research site in Puyallup, shows minilysimeters being weighed. The pots are weighed for daily water evaporation, rewetted, and returned to their “holes” in the turf. These scientific measurements under controlled procedures help researchers better understand low-water use.

plant should or should not be placed on a “low water-use” list. As has been seen time after time, it typically is not the plant that wastes water, but the person who is in charge of its care.

When establishing new lawns, turfgrass sod has been shown to require less water than seeding; the evaporative losses from bare soil are greater for seeded areas than they are for turf-covered soil beneath sod (Fig. 7).

Water “harvesting” and reuse is another water-conserving practice gaining greater use. It can be employed to conserve public water supplies and recharge groundwater supplies.

Historically, planners and designers have focused their efforts on moving rainwater and snowmelt away from a property as quickly as possible, giving little thought to the possible advantages of using that water for landscape purposes. Consider the fact that during a 1-inch rainfall, a 35-foot-by-60-foot roof (approximately 2000 square feet) will collect nearly 1250 gallons of water. Rushing this water to gutters and then sewers makes little sense when it could conceivably go into a system that could capture or distribute it across a landscaped area.

Another increasingly feasible source of additional landscape water is recycled or gray water. Some communities are installing dual water-delivery systems where one carries potable water for drinking, cooking, cleaning, and other general household uses and a second system delivers less thoroughly treated (but very safe) water for use on landscapes. On a small scale, some locales encourage collecting a home’s gray water (from clothes washers, etc., but not toilets) for use on landscapes.

After being applied to a landscape area, harvested, recycled, or gray water is either transpired by plants and evaporates into the atmosphere or finds its way into groundwater supplies after it has been cleansed by the plant’s root structure.



Figure 7. When establishing new lawns, turfgrass sod has been shown to require less water than seeding.

As science and technology continue to advance, new and better information, tools, and systems will become available to help people establish and maintain water-conserving and environmentally beneficial landscapes. An ongoing challenge will be keeping pace with these developments, sharing that information, and continually improving best management practices.

Just as we improve our health by using water to brush our teeth and to wash our bodies and clothes, applying water judiciously to a properly designed and installed landscape can improve our health and the general environment.

Efforts to eliminate landscape water use not only take away freedom of personal choice; they also bring many environmental, economic, and emotional drawbacks that could be more costly in the long run.

CROP WATER REQUIREMENTS

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Crop water requirements (CWR) refer to the amount of water required to compensate for evapotranspiration losses from a cropped field during a specified period of time. Crop water requirements are expressed usually in mm/day, mm/month, or mm/season, and they are used for management purposes in estimating irrigation water requirements, irrigation scheduling, and water delivery scheduling.

The concept of crop water requirements is intimately connected with crop evapotranspiration because both refer to the same amount of water. Nevertheless, there is some difference between them. Crop evapotranspiration represents the water losses that occur (i.e., a hydrological term), but crop water requirement indicates the amount of water that should be supplied to account for these losses (i.e., an irrigation management term). In fact, this amount of water corresponds to the effective irrigation water supply necessary to reach the maximum yield. Therefore, the estimation of crop evapotranspiration precedes the estimation of crop water requirements where the latter usually represents the values of crop evapotranspiration aggregated over some period of time. For management purposes, crop evapotranspiration should be transformed into crop water requirements to be used for estimating irrigation water requirements and later, management of the irrigation water supply. Then, crop water requirements are satisfied by the amount of irrigation water effectively supplied to the root zone and the effective precipitation.

The concept of crop water requirements has become important due to the development of large engineering works when it was necessary to estimate the water volumes to be supplied to newly irrigated areas. In general, it is possible to distinguish between crop water requirements for long-term planning, where an average

climate or a climate with a certain probability of occurrence can be used for a CWR estimate, and crop water requirements for real-time management, where climatic data of the ongoing season are applied.

The procedures for estimating crop water requirements correspond to those of crop evapotranspiration because both terms refer to the same amount of water. More details of measuring and estimating crop water requirements are given under the article on CROP EVAPOTRANSPIRATION of this Encyclopedia.

Example of an estimate of seasonal crop water requirements for tomato growing in the Mediterranean region based on a planting date of April 15th and where the average monthly reference evapotranspiration values are as follows: $ET_{O\text{April}} = 3.5$ mm/day; $ET_{O\text{May}} = 4.4$ mm/day; $ET_{O\text{June}} = 5.1$ mm/day; $ET_{O\text{July}} = 5.8$ mm/day; $ET_{O\text{August}} = 5.6$ mm/day, and $ET_{O\text{September}} = 4.2$ mm/day.

Calculation Procedure. Some data necessary for calculation are taken from the article on CROP EVAPOTRANSPIRATION of this Encyclopedia: the lengths of the growing stages are taken from Table 3 and K_c values from Table 4 of that article. The growing cycle of tomato can be presented graphically as in Fig. 1: K_c is fixed at 0.5 from April 15th to May 15th, then it increases linearly until June 24th when it reaches the value 1.15; K_c is fixed at 1.15 from June 24th to August 7th; then it decreases linearly to 0.8 on September 6th, the last day of growing cycle. Hence, the average monthly values of K_c are determined as follows: $K_{c\text{April}} = 0.5$; $K_{c\text{May}} = 0.57$; $K_{c\text{June}} = 0.94$; $K_{c\text{July}} = 1.15$; $K_{c\text{August}} = 1.04$; $K_{c\text{September}} = 0.835$.

Then, the crop water requirements are calculated for each month as

$$\begin{aligned} \text{CWR}_{\text{April}} &= K_{c\text{April}} * ET_{O\text{April}} * 15 \\ &= 0.5 * 3.5 * 15 = 26.2 \text{ mm/month} \end{aligned}$$

$$\begin{aligned} \text{CWR}_{\text{May}} &= K_{c\text{May}} * ET_{O\text{May}} * 31 \\ &= 0.57 * 4.4 * 31 = 77.8 \text{ mm/month} \end{aligned}$$

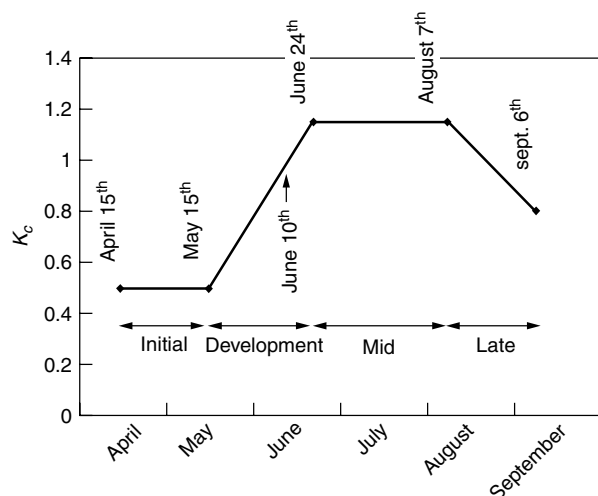


Figure 1. The variation of K_c for a tomato crop during the growing season.

$$\begin{aligned} \text{CWR}_{\text{June}} &= K_{c\text{June}} * ET_{O\text{June}} * 30 \\ &= 0.94 * 5.1 * 30 = 143.8 \text{ mm/month} \end{aligned}$$

$$\begin{aligned} \text{CWR}_{\text{July}} &= K_{c\text{July}} * ET_{O\text{July}} * 31 \\ &= 1.15 * 5.8 * 31 = 206.8 \text{ mm/month} \end{aligned}$$

$$\begin{aligned} \text{CWR}_{\text{August}} &= K_{c\text{August}} * ET_{O\text{August}} * 31 \\ &= 1.04 * 5.6 * 31 = 180.5 \text{ mm/month} \end{aligned}$$

$$\begin{aligned} \text{CWR}_{\text{September}} &= K_{c\text{September}} * ET_{O\text{September}} * 6 \\ &= 0.835 * 4.2 * 6 = 21.0 \text{ mm/month} \end{aligned}$$

The seasonal crop water requirements represent the sum of the monthly values, 656.1 mm/season.

Crop water requirements can also be estimated daily, provided that daily values of reference evapotranspiration are available. There, K_c values can be interpolated similarly, as in the previous example, or they can be estimated using more complicated approaches (e.g., the dual K_c approach).

Crop water requirements of one crop variety vary from one location to another due to variations in climatic parameters and, therefore, to different reference evapotranspiration, different planting dates and lengths of growing season, different K_c values, etc.

AGRICULTURAL WATER USE EFFICIENCY (WUE) AND PRODUCTIVITY (WP)

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Among the different sectors of society, agriculture represents the major water consumer. Rain-fed and irrigated agriculture uses water at various degrees of efficiency, resulting in different levels of productivity.

The general term *efficiency* (e) indicates the ratio of *output* to *input* of variables (Eq. 1) associated with *transformation* (e.g., of energy) or *transport* (e.g., of mass and energy) *processes*:

$$e = \frac{\text{output}}{\text{input}} \quad (1)$$

The *efficiency* term (e) has two peculiarities: (1) the units are nondimensional and (2) the values are always between the theoretical limits of 0 and 1. In water use, the efficiency (e_w) has water amounts (e.g., volumes, quantities, or depth) in both *input* and *output* variables.

The general term *productivity* (p) still indicates a ratio of output to *input* of variables, as in Equation 1, but these *output* and *input* variables are not associated with *transport processes* and might be associated, but not necessarily, with a *transformation process*. The consequent peculiarities of the *productivity* term (p) do not match those of the *efficiency* term (e), that is, the units are dimensional, and the theoretical limits of 0 and 1 are lost. In water use, the productivity term (p_w) has water

amounts (e.g., volumes, quantities, or depth) only in the *input*. The *output*, instead, generally takes the form of quantities produced: physical (e.g., dry matter or final yield of crops, or meat or dairy products from animal husbandry, etc.), economic (e.g., gross income, net revenue, etc.), or any other form.

The efficiency term e_w has to do more with the way water is used by the system, whereas p_w has to do more with the outcome from the system per unit of water it consumes. Nevertheless, it turns out that a change in e_w is reflected in a change in p_w , implying that there is a link between e_w and p_w , though the reverse is not the case. Moreover, though p_w cannot be strictly defined by Equation 1, it still resembles a sort of efficiency of the system to produce a given *output* for a given *input*. The undefined degree of commonality and distinction between e_w and p_w has led to some confusion in investigating agricultural water use.

A GENERAL FRAMEWORK FOR AGRICULTURAL WATER USE EFFICIENCY AND WATER PRODUCTIVITY

Figure 1 illustrates a suitable framework for identifying the *efficiency* (e_w) and the *productivity* (p_w) components within the domain of agricultural water use. The framework distinguishes irrigated from rain-fed agriculture and splits the overall path that water “travels” to produce a given yield into single segments of particular relevance and specific boundary conditions.

In irrigated agriculture, the source of water can be a reservoir (e.g., a dam) from which water will be conveyed and delivered to the farm. Applying Equation 1 to the whole path from the reservoir to a cropped field at farm level, will express the *efficiency* (e_w) in water transport where the *input* is the outflow from the reservoir (Res_w_out) and the *output* is the water effectively stored in the soil root zone ($Root_zone_w$). The whole path can

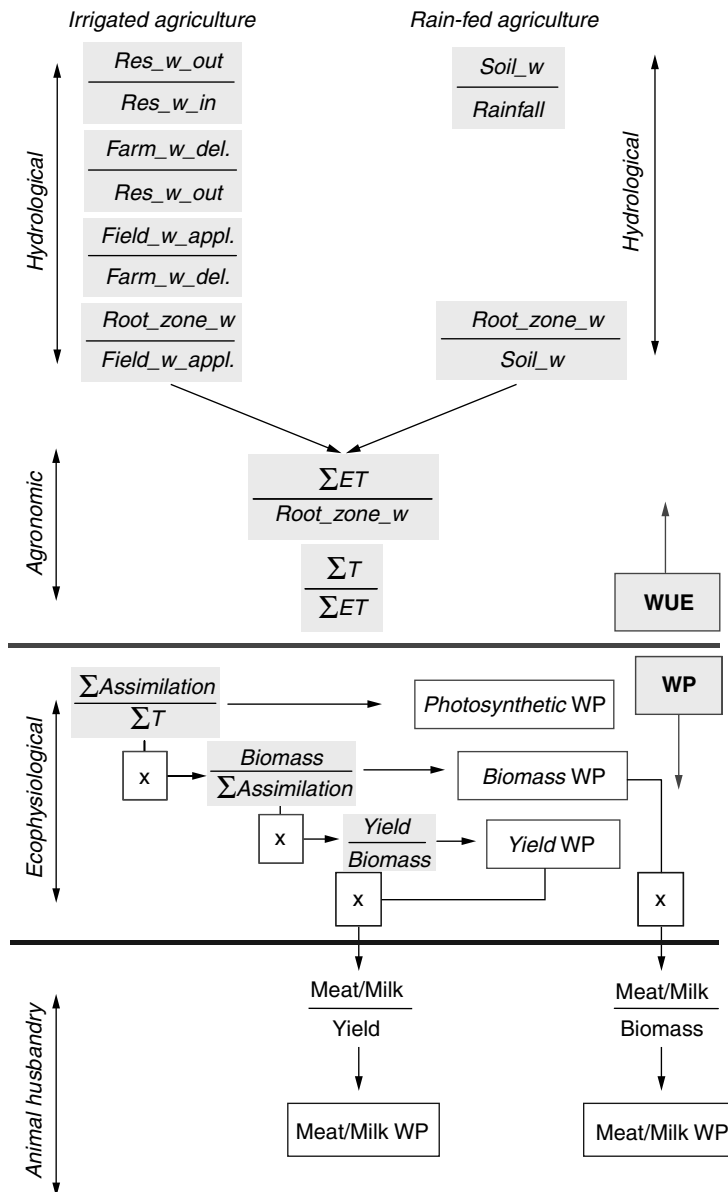


Figure 1. Suggested framework for analyzing components of water use efficiency and water productivity in agricultural crop production and animal husbandry. The overall path “traveled” by water is split into different segments where appropriate analysis can be performed. See text for explanation.

be split into different segments (e.g., from the dam to the collective irrigation system; from the collective system to the hydrant of a single farm, from the hydrant to the cropped field where an irrigation system is present, and from the outlets of the irrigation system into the root zone of the crop being irrigated) to investigate the single component efficiencies and the overall efficiency obtained from the product of the single-segment efficiencies.

In rain-fed agriculture, rainfall represents the source of water, and the ratio of the water stored in the soil (*Soil_w*) to the water from the rain (*Rainfall*) represents the rain storage *efficiency*. Out of all the water stored in the soil (*Soil_w*), only a portion may be available in the soil explored by crop roots (*Root_{zone_w}*), and this represents the available water *efficiency*. All efficiencies mentioned so far can be qualified as *hydrological*.

After the final allocation of water in the *root zone* (whether irrigated or rain-fed), water follows a common pathway for the crop production process: it enters the plants and evapotranspires into the atmosphere. We can identify two additional efficiencies that may be qualified as *agronomic*: one is expressed by the ratio of the amount of water used by evapotranspiration (ΣET) to the water stored in the soil root zone (*Root_{zon_w}*); the other is expressed by the ratio of the amount of effectively transpired (ΣT) and the total amount of evapotranspired water (ΣET). This last segment of the water path is essential because nontranspired water is not considered productive to the crop. Thus, a distinction has to be made between soil evaporation (*E*), which is a loss, and crop transpiration (*T*).

At this point, the water has finished its trip from the source, all the way through the system, passing through the soil, entering the plant, and ending up into the atmosphere, beginning as a liquid phase and ending up as a vapor phase. So far, all the expressions of *efficiency* for the different segments (either those pertaining to the *hydrological* pathway or those pertaining to the *agronomic* pathway) properly match the definition of Equation 1. because they are nondimensional and have theoretical limits between 0 and 1.

Examining the remaining segments of the framework reported in Fig. 1, we next enter the *physiological* (or better, *ecophysiological*) domain of the agricultural system where the term *efficiency* loses its rigorous meaning and it is more appropriate to talk in terms of *water productivity* (p_w). Three major expressions for p_w can be identified: (1) *photosynthetic water productivity* (*photosynthetic p_w*); (2) *biomass water productivity* (*biomass p_w*); and (3) *yield water productivity* (*yield p_w*).

Photosynthetic p_w is defined as the ratio of leaf net assimilation to leaf transpiration (A/T) and varies among species according to their photosynthetic pathways. These species can be grouped into three major types: the C_3 , C_4 , and CAM plants. All three types ultimately rely on the Calvin photosynthetic carbon reduction cycle for their CO_2 assimilation, but some additional features characterize those groups that make the *photosynthetic p_w* of CAM plants higher than that of C_4 plants, which in turn is higher than that of C_3 plants, in comparable environmental conditions. Values of *photosynthetic p_w* may

also vary according to the nutritional status of the leaves and their age. Variable environmental conditions (e.g., radiation regime, dryness of the atmosphere, etc.) may also affect the values of *photosynthetic p_w* . Under equal ambient conditions, all species belonging to the same group type have similar *photosynthetic p_w* values.

Biomass p_w is defined as the ratio of cumulative above-ground biomass (biomass) of a crop canopy to cumulative transpiration (ΣT) by the crop. The relationship between cumulative aboveground biomass and ΣT is generally quite strict. The slope of the relationship may vary from year to year and from location to location, depending on the different environmental conditions and the biomass composition of the various species. When ΣT is normalized for a reference evaporative demand of a given climatic environment and the nutritional conditions are similar, many crop species follow the same slope and group into the same types as indicated for the *photosynthetic pathways p_w* (C_3 , C_4 , and CAM) and the main production compounds (e.g., carbohydrates, proteins, or lipids).

Yield p_w is defined as the product of *biomass p_w* times the harvest index (HI, i.e., the ratio of the marketable yield to aboveground biomass). *Yield p_w* values are variable among species and cultivars, depending on the values of HI, which in turn depend on the partitioning of assimilates in the various plant organs (e.g., leaves, stem, grains). Of course, HI can also vary greatly as a consequence of the environmental conditions where the plant is growing (e.g., water stress, nutritional deficit, etc.).

The water productivity concept can be extended to animal production systems when the crop's biomass and/or yield are used as feed. Then, the product of biomass or yield water productivity is multiplied by the factor of conversion of the feed into animal production (dairy product or meat). The ratio approach can be extended throughout the system up to the final step of interest (Fig. 1).

Furthermore, when the amount of product (the nominator of the productivity ratio) is expressed as gross income (e.g., multiplying it by the price) or as net income (e.g., deducting production costs), the ratio can represent the *economic water productivity* (*economic p_w*), better indicated as *water return* (r_w) having the unit of "currency" (dollars, euro, etc.) per unit amount of water (volume, quantity, or depth).

LARGE AREA SURFACE ENERGY BALANCE ESTIMATION USING SATELLITE IMAGERY

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INTRODUCTION

Many activities such as irrigation and agricultural scheduling, water resources planning, weather forecasting, atmospheric circulation modeling, energy resources management and development, and even health related issues involve studying the surface energy balance (EB) of

the earth. Photosynthesis and other metabolic processes in green plants, evaporation from free water and soil surfaces, wind effects, atmospheric and oceanic circulation, and many other phenomena are mainly functions of solar radiation. In other words, they are functions of surface EB, in which solar radiation is the most important driving force.

Due to nonuniform properties of the earth's surface (albedo, heat capacity, soil cover, etc.) and also due to temporal variation of the earth's surface properties, the earth's surface phenomena must be monitored on spatial and temporal scales. Due to spatial variations, ground-based monitoring of the earth surface can cover only small areas because of economic restrictions.

Generally, installing a dense monitoring network is not feasible or it is technically impossible. For example, first-order climatology stations may be 100 km apart. But variations in soil texture, soil cover, topography and microclimate may occur on a much smaller spatial scale. On the other hand, interpolation/extrapolation of point observations may be insufficient due to weak ground coverage. And, the need for distributed models increases.

Satellite imagery provides a large view of the earth's surface. Such a spatially continuous, but time discrete, data acquisition system provides a suitable framework for distributed modeling, provided that the phenomenon to be modeled has spectral behavior.

All features on or near the earth's surface, including soil surface, water bodies, vegetation cover, and near surface atmosphere, exhibit spectral characteristics. Therefore, remote sensing can monitor the earth's surface directly or indirectly. During the past two decades, several satellites were launched with a large variety of sensor capabilities. Many articles in the literature report using remote sensing to study the energy partitioning of the earth's surface. In this respect, Bastiaanssen (1) developed a model for energy partitioning using a remote sensing algorithm named SEBAL (surface energy balance algorithm for land). Bastiaanssen et al. (2) used SEBAL to model evapotranspiration on a catchment scale in Spain, Niger, China, and Egypt. Bastiaanssen (3) also tested the model in Turkey. Silberstein et al. (4) correlated the surface effective temperature with the satellite-derived surface temperature to form the energy balance on a catchment scale.

Remote sensing is being applied in a large number of investigations, such as Xin and Shih (5), Laymon et al. (6), and Biftu and Gan (7) to model surface evaporation and canopy evapotranspiration. Bastiaanssen et al. (8) introduced the possibilities of applications in agriculture, and Arasteh (9) developed a model to estimate evaporation from the Hamoon Wetlands called HRSE (Hamoon remotely sensed evaporation) within an area of 4000 km² on the Iran–Afghanistan border using NOAA-AVHRR images.

The advantages of using remote sensing in hydrologic studies have been stated by Prince (10) and Dubayah et al. (11) as follows:

1. large aerial coverage
2. higher spatial resolution than ground-point observations

3. availability of long-term records from area without ground observations
4. possibility of change detection

In this article, the EB is studied using remotely sensed data and some common methods are introduced. Then, the EB components of the Hamoon Wetlands are described as a case study.

ENERGY BALANCE

All climatic and hydrologic phenomena in the atmosphere and on the earth's surface are affected by the EB of the surface. The EB is one of the main subjects in climatology, hydrology, civil and traffic engineering, biology, and health. EB is the balance between the inward and outgoing energy components of the surface.

Understanding the behavior of the terrestrial hydrosphere, biosphere, and climatic systems and their possible changes requires knowledge of the nature of the EB (11). The EB of a surface is the radiative transfer of energy through the top of the earth's surface and the molecular energy transfer into the soil matrix or free waterbody (12). Fig. 1 shows the components of the EB.

As shown in Fig. 1, the EB has several components. The net radiation flux of the surface, R_n , is the algebraic sum of the following components:

$$R_n + A - S - G - \lambda E - \Delta Q = 0, \quad (1)$$

where A , S , G , λE , and ΔQ are advective and convective sensible heat, soil heat, latent heat fluxes and change in heat storage, respectively. All of the fluxes have the dimension of MT^{-3} with units such as $MJ/m^2\text{-day}$. In Eq. 1, all sources of energy (R_n and A) are positive values, and sink terms (S , G , λE , and ΔQ) are negative.

There is also a radiation balance at the surface, in which the net radiation is calculated from the algebraic sum of the inward and outward radiations (Eq. 2).

$$R_n = R_s \downarrow - R_s \uparrow + R_l \downarrow - R_l \uparrow, \quad (2)$$

where R_s and R_l are short- and long-wave radiation, respectively; $\downarrow$ and $\uparrow$, represent inward and outward radiation to and from the surface.

Using Stephan's law, Eq. 2 changes to (6,13):

$$R_n = (1 - \alpha)R_s + C\sigma(\varepsilon_{sky}T_a^4 - \varepsilon_{surface}T_s^4) \quad (3)$$

where α is surface albedo; ε_{sky} and $\varepsilon_{surface}$ are the emissivities of the atmosphere and the surface, respectively; C is the cloudiness factor; σ is the Stephan–Boltzman constant ($4.903 \times 10^{-9} \text{ MJ/K}^4\text{-m}^2\text{-day}$); and T_s and T_a , are surface and near surface atmosphere temperatures (K).

Other components of EB, S and λE , may be determined by aerodynamic resistance relations (6,13,14) and G can be modeled on the basis of heat conduction theory (12):

$$S = \frac{\rho C_p}{r_a}(T_s - T_a) \quad (4)$$

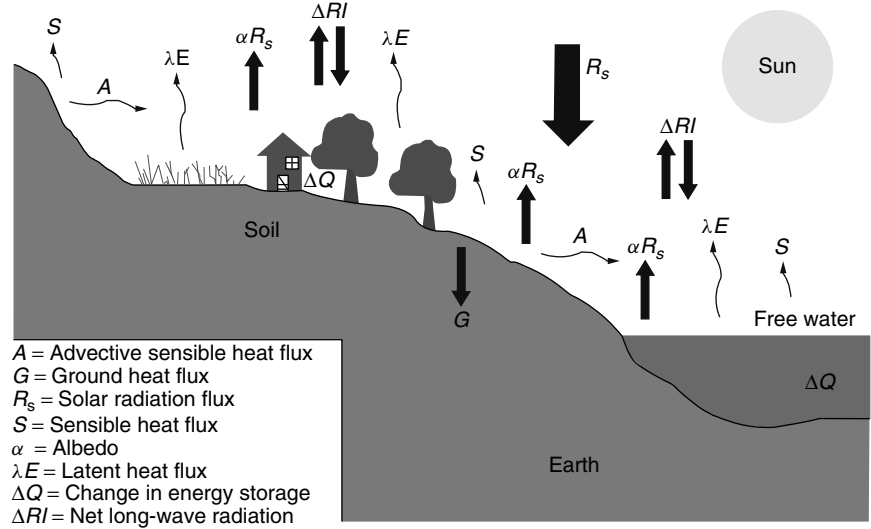


Figure 1. Schematic components of the EB.

$$\lambda E = \frac{\rho C_p e_s (T_s) - e_a}{\gamma \frac{r_a + r_s}{r_a + r_s}} \quad (5)$$

$$G = dC_s \frac{dT}{dt} \quad (6)$$

where ρ is air density (kg/m^3); C_p is the specific heat of air (MJ/kg-K); C_s is the soil matrix or waterbody heat capacity ($\text{MJ/m}^3\text{-K}$); γ is the psychrometric constant; r_a and r_s are aerodynamic and surface resistances, respectively (s/m); e_s and e_a are saturated vapor pressure at T_s and actual air vapor pressure, respectively (kPa); d is the depth of soil or water (m); and dT/dt is the change in temperature per unit time.

Equations 3–6 have some variables that can be determined by remote sensing and others must be obtained from ground observations. The received solar radiation, R_s , is introduced in several texts on the basis of the earth–Sun geometric relationships (15). Determination of surface temperature and near surface air temperature, dew point, precipitable water in atmosphere column, vapor pressure deficit, and albedo are reviewed by Cracknell, (16), Prince et al. (10), Dubayah et al. (11), Menenti (17), Granger (18), and Schmugge et al. (19).

REMOTELY SENSED EB

Satellite remote sensing is a collection of sciences, technologies, and methods of inferring the earth's surface parameters by measuring the electromagnetic radiation of the surface (18). Visible (VIS), near infrared (NIR) and thermal infrared (TIR) channels are used for EB processing (16,17,19).

Some primary surface parameters such as surface albedo, surface temperature, air temperature, surface roughness, land cover, and land use may be provided by remote sensing. In the following, satellite data acquisitions are introduced.

Surface Albedo

Reflected directional radiance in VIS and NIR channels (0.4 to $2.5 \mu\text{m}$) are being used to determine surface albedo (16–18). To determine surface albedo, Gutman (20) and Csiszar and Gutman (21) introduced the following methodology:

1. Normalizing VIS and NIR channels reflectances with respect to solar zenith angle (SZA):

$$\begin{aligned} (\rho_1)_{\text{TOA}} &= R_1 \cos(\text{SZA}) \\ (\rho_2)_{\text{TOA}} &= R_2 \cos(\text{SZA}) \end{aligned} \quad (7)$$

where R is reflectance; ρ is normalized reflectance; SZA is solar zenith angle (radians); 1 and 2 indexes represent VIS and NIR channels, respectively; and TOA is top of atmosphere.

2. Converting VIS and NIR narrowband reflectance to a broadband reflectance by a linear relation to determine top of atmosphere (TOA) reflectance:

$$(\rho)_{\text{TOA}} = a + b(\rho_1)_{\text{TOA}} + c(\rho_2)_{\text{TOA}} \quad (8)$$

where a , b , c are regression coefficients.

3. Correcting TOA reflectance for anisotropy to determine TOA albedo:

$$(\alpha)_{\text{TOA}} = (\rho)_{\text{TOA}} (\text{ARF})^{-1} \quad (9)$$

where ARF is the anisotropic reflectance factor.

4. Modifying TOA albedo for atmospheric effects to determine surface albedo:

$$\alpha = f[(\alpha)_{\text{TOA}}, \text{SZA}, \text{AWV}, \text{AA}] \quad (10)$$

where α is albedo (%), AWV is atmospheric water vapor, and AA are atmospheric aerosols. Song and Gao (22) gave calibrated nonlinear relations between

coefficients b and c and NDVI (normalized difference vegetation index) or SAVI.

$$\text{NDVI} = \frac{\text{NIR} - R}{\text{NIR} + R} \quad (11)$$

$$\text{SAVI} = 1.5 \left(\frac{\text{NIR} - R}{\text{NIR} + R + 0.5} \right) \quad (12)$$

where R is the red channel of the spectrum.

Instead of the linear relation of Equation 8 Granger (18) emphasized that the AVHRR-NIR channel could be used individually to determine the out of atmosphere reflectance and the corresponding albedo.

Surface Temperature

Surface temperature is the result of the thermodynamic equilibrium between the energy condition in the near surface atmosphere, soil surface, and subsurface soil. It is also a function of surface emissivity (19). The first step in inferring surface temperature by remote sensing is to determine the brightness temperature, T_B , of TIR channels. The T_B at satellite temperature is the temperature of a blackbody and can be determined by solving Planck's spectral distribution function for blackbody temperature (16):

$$R(T_B) = \frac{\int_{\nu_1}^{\nu_2} E(\nu, T_B) \Phi(\nu) d\nu}{\int_{\nu_1}^{\nu_2} \Phi(\nu) d\nu} \quad (13)$$

where R is radiance ($\text{MJ/m}^2\text{-Sr-cm}$); T_B is brightness temperature (K); E is Planck's radiation energy of a blackbody ($\text{MJ/m}^2\text{-Sr-cm}$); Φ is spectral response function; and ν is wave number equal to the inverse of wavelength (cm^{-1}).

Spectral radiance, R , is determined from the sensor calibration equation on the basis of the image digital number. Equation 13 must be solved numerically for energy E . Then the brightness temperature, T_B , is determined by solving Planck's law (16):

$$T_B = \frac{C_2 \nu}{\ln \left(1 + \frac{C_1 \nu^3}{E} \right)} \quad (14)$$

in which $C_1 = 1.1910659 \times 10^{-5} \text{ MJ/s-m}^2\text{-Sr-cm}^{-4}$ and $C_2 = 1.438833 \text{ [cm-K]}$.

To determine surface temperature, there are several methods classified in three major categories:

1. Ground observation based calibration
2. Atmospheric effects back-modeling
3. Atmospheric effects omission (two-look method and multichannel method or split window)

Cracknell (18) has described these methods in detail. The split window method for determining surface temperature is the most common (10,18,23,24).

Air Temperature

Ground observation calibration methods can be used to determine air temperature. Air temperature can also, be derived from the thermal vegetation index (10,11) where the surface temperature of a closed canopy is assumed equal to its adjacent air temperature.

Sky and Surface Emissivities

Surface and near surface atmosphere emissions are related to surface indexes such as NDVI, SAVI, etc. (6,25):

$$\varepsilon_{\text{surface}} = a_1 \cdot \text{NDVI} + a_2 \quad (15)$$

$$\varepsilon_{\text{surface}} = b_1 \cdot \ln(\text{NDVI}) + b_2 \quad (16)$$

where a_1 , a_2 , b_1 , and b_2 , are regression coefficients.

Net Radiation, Sensible and Latent Heat Flux Densities

According to Eq. 3, from the earth-Sun geometry and satellite data acquisition, using previously mentioned methods, one can determine R_n . Using satellite derived T_s and T_a and ground based data, other energy components of the surface of Eqs. 4–6 can be determined. In this respect, surface and aerodynamic resistances and surface roughness length are related to LAI (leaf area index) and consequently to remote sensing by indexes such as NDVI or SAVI (3,25):

$$Z_{0m} = \exp(a' \cdot \text{NDVI} + b') \quad (17)$$

$$\text{LAI} = a'' \cdot \ln(b'' + \text{NDVI}) \quad (18)$$

$$\text{LAI} = a''' \cdot \exp(b''' \cdot \text{NDVI}) \quad (19)$$

$$r_s = f(\text{LAI}) \quad (20)$$

$$r_a = f(Z_{0h})$$

where Z_{0m} and Z_{0h} are the surface roughness length due to turbulence and heat, respectively; r_a and r_s , are aerodynamic and surface resistances; and a' , a'' , a''' , b' , b'' , and b''' are regression coefficients.

Case Study

To show an example of a remotely sensed EB application, the study by Arasteh (9) on Hamoon Wetlands is summarized as follows:

The Hamoon Wetlands located between Iran and Afghanistan form a complex hydraulic-hydrologic system of three neighboring wetlands with an average water area of 4000 km^2 in wet years. The wetlands are transboundary and affect all human activities in the region (Fig. 2).

Determining an estimate of evaporation from the free water surface of the Hamoon Wetlands was the main objective of the study by Arasteh (9). For this, it was required to study remotely sensed EB. Arasteh (9) used energy partitioning to fulfill this objective. Using his algorithm with the NOAA-AVHRR-14 image of May 10, 1998, the net radiation, sensible heat, and latent heat of the Hamoon surface are shown in Fig. 3.

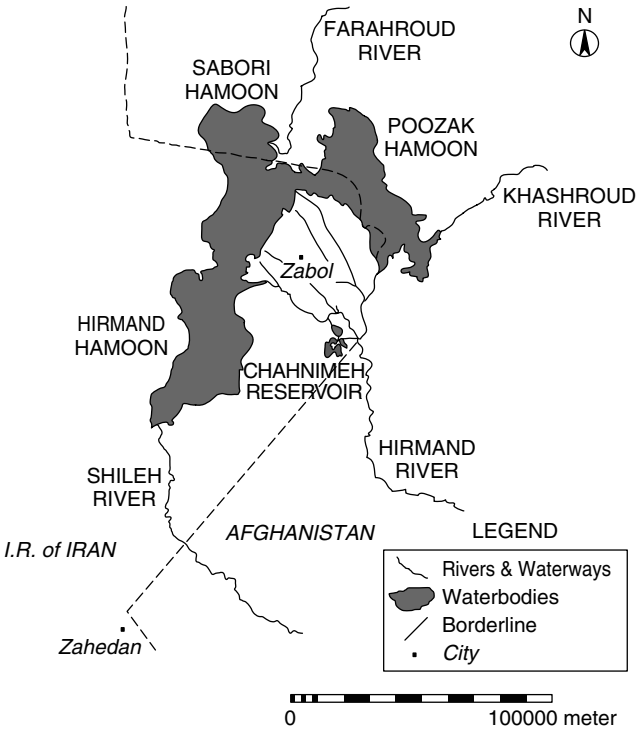


Figure 2. Hamoon Wetlands region between Iran and Afghanistan.

DATA ACCURACY

To use each method, it is necessary to consider its accuracy. Therefore, it is very important to consider the question of the accuracy of remotely sensed EB.

Considering Eqs. 1 and 3–6, there are various variables with their own sources of error. Cracknell (16) reported a variation of 0.5 to 1.4 K in the sea surface temperature for only a 2% change in surface emissivity. Menenti (17) reviewed the literature and reported errors of less than 3% in latent heat and obtained an average error of

15% for this component. Dubayah et al. (11) stated that it is not possible to estimate downwelling long-wave radiation with accuracy less than 20 W/m² using AVHRR data.

Finally, it can be concluded that the accuracy of a remotely sensed EB is related mainly to the spatial resolution of images, the spectral resolution (number of channels and bandwidth for each channel), the accuracy of ground-based observations, and the range of applicability of calibrated equations.

SOURCES OF SATELLITE EB

Carleton (26) summarized some sources of satellite based EB (Table 1). He emphasized METEOSAT, GMS, and ERBS satellite systems as the most important sources of global coverage of EB. The ERBS is a professional system. The Earth Radiation Budget Experiment (ERBE) included ERBS and NOAA 9–10 platforms.

Table 1. Some Satellite Systems Related to EB Generation^a

Variable	Spectral Region	Sensor	Satellite	Type of Satellite
Surface albedo	VIS	SR	NOAA	Polar orbiter
	NIR	AVHRR	NOAA	Polar orbiter
		VISSR	GOES	Geostationary
Surface solar irradiance	VIS	SR	NOAA	Polar orbiter
		VISSR	GOES	Geostationary
Outgoing long-wave Radiation	TIR	SR	NOAA	Polar orbiter
		AVHRR	NOAA	

^aReference 26.

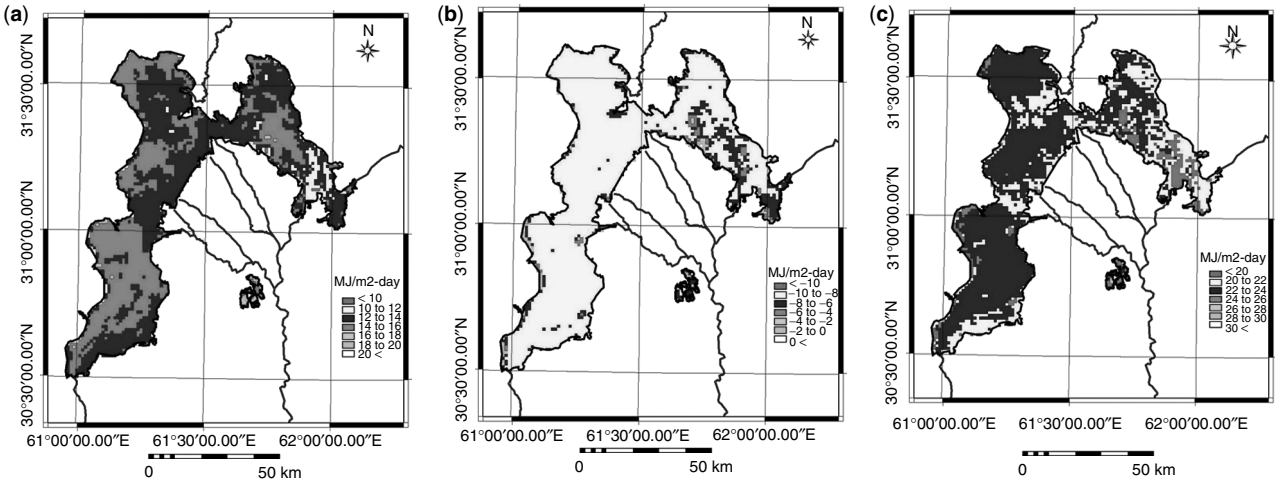


Figure 3. Energy partitioning for Hamoon Wetlands using the Arasteh (9) algorithm; (a) net radiation, (b) sensible heat, and (c) latent heat.

RELATED INTERNET SITES

1. <http://www.saa.noaa.gov/nsaa/products/>.
2. <http://www.osdpd.noaa.gov/PSB/EPS/RB/>.
3. <http://orbit-net.nesdis.noaa.gov/goes/gcip/>.
4. http://www.cdc.noaa.gov/~map/maproom/text/Climate_Pages/olr.shtml.
5. <http://poes.nesdis.noaa.gov/climate/>.
6. <http://www.ncdc.noaa.gov/>.
7. <http://www.eurometeo.com/>.

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SOIL EROSION AND CONTROL PRACTICES

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INTRODUCTION

Soil erosion is the removal of soil or other slope-forming materials by erosive agents, namely, water and wind. Accelerated soil loss, in excess of the rate of natural soil erosion, is perpetuated when surface vegetation is compromised or removed. It is therefore more likely to occur wherever land is used by humans

for agricultural production, infrastructural construction, mineral abstraction, or recreation, and where measures for soil protection are not in place.

Soil erosion involves a two-phase process of soil detachment and transport, both of which require an energy input. Much of the energy of erosion derives from water or wind flow over the soil surface or is provided by gravity. Although reducing the energy of erosive forces is the most effective way of controlling soil erosion, one consequence of reducing energy may be the deposition of any material already eroded. As eroded sediment is often enriched with nutrients, organic matter, and contaminants preferentially adsorbed onto the sediment surface, sediment deposition may allow contaminant release into the environment. It is necessary, therefore, to reduce erosive energy at the source areas of sediment rather than after the sediment has been detached and transported.

SOIL EROSION

Water Erosion

Rainsplash (detachment and removal of soil particles and aggregates at the soil surface by raindrop impact) is an important soil-detaching agent. The energy available for erosion is related to raindrop size and velocity. Raindrops have greater kinetic energy than the same volume of runoff and are thus potentially more erosive, although most raindrop energy is expended in soil detachment, leaving little energy available for sediment transport. Rainsplash plays a further role in slope erosion through the detachment, dispersion, and redeposition of fine soil particles on the soil surface. Soil pores are blocked by the redeposited particles, resulting in a surface seal, a reduction in infiltration, and an increase in erosive runoff.

Rainsplash constitutes only a negligible proportion of total soil loss except where runoff is not generated, where it can be the dominant process of sediment removal (1). Where runoff does occur, particle suspension is greatly enhanced by raindrop impact (2) and rainsplash operates as a "feeder" mechanism to rill and runoff erosion. Runoff may also present a buffer between falling raindrops and the soil surface, so that detachment by rainsplash can be limited by runoff. This depends on water depth and raindrop size (3).

Runoff in unchanneled form has a relatively low kinetic energy and, while able to transport soil particles thrown into the flow by raindrop impact, is not able to detach new material. Hortonian runoff (when rainfall intensity exceeds the infiltration capacity of the soil) depends critically on rainfall intensity and amount and soil infiltration capacity. Saturation runoff (when rainfall amount exceeds soil storage capacity) depends on rainfall intensity, subsoil permeability, and the porosity and permeability of the topsoil.

Runoff channeled into rills or gullies has higher kinetic energy and is a much more powerful agent, able to detach and transport soil particles of all sizes, including gravels and small stones. Its considerable erosive power can account for up to 78% of the total material transported within an erosion event.

Wind Erosion

The main factor affecting wind erosion is the velocity of moving air. Because of the roughness imparted by vegetation, soil, and other obstacles, wind speeds are lowest nearest the ground surface. Above this, wind speed increases exponentially with height. The movement of soil particles can be related to critical shear velocity, which is the velocity of wind required to dislodge soil particles from the ground surface and is dependent on soil particle size and soil cohesiveness. Very high shear velocities are required to initiate soil particle movement by wind, and this energy may be translated to adjacent soil particles at the same time, thus causing a chain reaction of motion. Sediment-laden air also acts to detach and mobilize other soil particles by bombarding or impacting on the soil surface. Once in motion, soil particles are transported by wind in suspension (in the air), as surface creep (rolling along the ground surface), and as saltation (particle movement in jumps) (4).

Rates of Soil Erosion

Rates of soil erosion are normally expressed in terms of the mass or volume of soil removed from a unit area of land per unit of time (t or $m^3 \cdot ha^{-1} \cdot y^{-1}$). Typical annual rates of erosion on hillslopes under forest or natural grassland are 0.01 – $0.5 t \cdot ha^{-1}$ in areas of moderate slope steepness and 0.1 – $5 t \cdot ha^{-1}$ on steeper slopes. These rates are generally considered low and representative of erosion as a natural geomorphological process. On agricultural land, annual rates are much higher, ranging from 0.1 to $200 t \cdot ha^{-1}$, reflecting the poorer protection afforded by various crops. Arable land is particularly vulnerable to erosion seasonally when the soil is bare between ploughing and crop emergence and also from emergence until the crop canopy closes. Grazing land is vulnerable if overgrazing removes the vegetation cover. Even higher rates of erosion have been recorded on construction sites, often reaching 1000 or even $5000 t \cdot ha^{-1} \cdot yr^{-1}$.

Since erosion is a natural process, it is implicit that it cannot be prevented. Erosion can, however, be controlled at an acceptable rate, which should be no faster than the rate at which new soil forms. As little is known about rates of soil formation, except that they are extremely slow, a value of $1 t \cdot ha^{-1}$ (equivalent to $0.1 mm \cdot y^{-1}$) is sometimes accepted as an average annual rate for temperate areas. A more pragmatic approach is to consider the environmental damage that can occur if no control measures are in place (Table 1). Decisions can then be made about the level of damage tolerable relative to the costs of preventing that damage.

Impacts of Soil Erosion

Soil is a valuable resource *in situ* but when eroded by wind and water it can cause significant impacts both on and off site. The on-site implications of soil erosion include declines in cultivable soil depth, available soil moisture, soil organic matter, nutrient content, soil fertility, and soil structure as well as large-scale losses in productive area through gullying or mass movement (4). These on-site effects are particularly important on agricultural land:

Table 1. Rates of Erosion at a Field Scale and Associated Levels of Damage from Individual Storm Events

Erosion Rate, t·ha ⁻¹	Class	Indicators
<1	Natural	No surface evidence of erosion; land well-protected by vegetation cover (90–100% ground cover); very low sediment concentrations in runoff water, which appears clear.
1–2	Very slight	No surface evidence of erosion; land protected by vegetation cover (70–90% ground cover); low sediment concentrations in runoff water, which is clear to slightly cloudy.
2–5	Slight	Some crusting of the soil surface; localized wash, small channels (rills) every 50–100 m spacing; 30–70% ground cover; slight risk of pollution if runoff discharges directly into water courses.
5–10	Moderate	Evidence of surface wash; rills spaced every 20–50 m; exposed tree roots mark level of former soil surface; slight to moderate surface crusting; 30–70% ground cover; risk of pollution if runoff discharges directly into water courses.
10–50	High	Continuous network of rill channels every 5–10 m or deeper channels (gullies) every 50–100 m; surface crusting of soils over large area; <30% ground cover; danger of pollution and sedimentation downstream.
50–100	Severe	Land scarred by continuous network of rills every 2–5 m or gullies every 20 m; sediment splays of coarse material; bare soil; siltation of water bodies; damage to roads by erosion and sedimentation; risk of muddy floods.
100–500	Very severe	Land heavily scarred by continuous network of channels with gullies every 5–10 m; soil heavily crusted; bare soil; severe siltation, pollution, flooding, and eutrophication problems downstream.
>500	Catastrophic	Extensive network of rills and gullies; large gullies (>100 m ²) every 20 m; most of original soil surface removed; severe damage from erosion, pollution, flooding, and sedimentation downstream.

annual agricultural damage costs due to erosion have been estimated at £264 million in the United Kingdom alone (5). The net effect of these impacts is a loss in productivity, which both restricts what can be grown and results in increased expenditure on fertilizers and irrigation to maintain yields. Later this threatens food production and may ultimately lead to land abandonment, with a concurrent decline in the value of land as it changes from productive farmland to wasteland (4). A further consequence of this is the unsustainable exploitation of land resources elsewhere.

The deposition of eroded sediment reduces the capacity of rivers and drainage ditches, enhances flooding, blocks irrigation canals, and shortens the design life of reservoirs (4). Deposited sediment also represents a risk to infrastructure. Chemicals, including fertilizers and pesticides, and nutrients such as nitrogen (N) and phosphorus (P) deposited with eroding soil may have detrimental effects on water quality for supply, recreation, and fisheries. Further impacts on aquatic ecosystems include increases in turbidity reducing light penetration and O₂ availability, thereby reducing plant growth and key biological activities such as photosynthesis; submergence and physical damage to floodplain and in-stream plants; and clogging of fish spawning gravels.

CONTROLLING SOIL EROSION

Measures to control erosion rely on reduction of the energy of erosive forces. Reducing the rate at which soil is detached and transported involves increasing interception and infiltration of rainfall and runoff; reducing runoff volume, velocity, and energy; decreasing the area over which runoff occurs; increasing soil resistance; and increasing soil aggregate stability. The practical measures available to achieve these conditions

include mechanical, agronomic, and soil management methods of erosion control.

Mechanical Methods of Erosion Control

Mechanical methods of soil conservation are useful in the control of soil erosion by water. They include terraces, constructed on hill slopes to control soil erosion by reducing slope length and slope steepness, thus limiting the energy of runoff to detach and transport soil material (6). There are many different types of terrace but generic types include the channel and bench terrace. Channel terraces are suited to slopes no steeper than 7° (12%) (6) and consist of a cut channel and filled ridge (constructed from the cut material) constructed across the slope. The distance between consecutive terraces downslope is influenced by site conditions, with greater distances between terraces on gentler slopes and shorter terrace intervals on steeper slopes.

Bench terraces are steplike structures constructed across a slope to prevent erosion by dissecting the hill slope into smaller, shorter slope components, each with a level or near-level bench. The dimensions of the benches depend on slope steepness, the desirable width of cultivable land, and the acceptable height of the steep wall (riser) of the terrace, which is often related to soil depth. As construction of the level bench requires oversteepening of the terrace riser, this near-vertical wall must be supported with stones, masonry, or timber to prevent collapse.

In channel terrace systems, runoff is discharged into a channel constructed at a gentle gradient across slope to ensure that erosion within the channel itself does not occur. The channels are usually kept free of vegetation, so that any accumulated sediment can be excavated each year. From there, flow is directed to lined waterways, which are designed to cope with increased volume and velocity of runoff without causing

erosion. These waterways, ideally located in natural depressions to minimize the need for earth moving, are lined with vegetation, stones, or geotextiles with high shear resistance to minimize the risk of erosion.

Agronomic Methods of Erosion Control

Water Erosion Control. Agronomic methods involve the use of vegetation to control soil erosion and are highly effective (7). A plant canopy intercepts rainfall, changing raindrop size distribution and reducing fall velocities, so that kinetic energy available for erosion via raindrop impact is reduced. Plant stems distort and impart roughness to runoff paths, reducing flow velocity and thus kinetic energy to detach and transport soil material. Plant roots encourage infiltration through the soil profile and thus reduce the risk of runoff generation. Lateral plant roots also increase the resistance of soil against the shearing forces of runoff.

Agronomic methods of erosion control include ensuring maximum cover on the soil surface, especially during high rainfall periods, to protect the soil from erosive raindrops and maintaining high stem and root densities to reduce runoff volumes, velocities, and energy. These can be achieved through cover cropping, high density planting, crop rotations, mulching, and erosion control geotextiles. A geotextile is any permeable textile material that is an integral part of a constructed project, structure, or system. It may be made of synthetic or natural fibers. Geotextiles have many applications in land and water management, but those used for erosion control are usually in the form of two- or three-dimensional mats, webs, nets, grids, or sheets.

Natural fiber geotextiles are biodegradable and designed to be laid over the surface of the soil to provide protection against erosion until a vegetative cover is established: in this way, geotextiles act as simulated vegetation. Artificial fiber geotextiles are designed to give permanent protection to the soil surface and are usually buried to reinforce the soil both before and after vegetation establishment. Surface-laid, natural fiber geotextiles are more effective at controlling soil erosion by raindrop impact because they provide good surface cover, high water absorption, and a rough surface on which water can pond. They also reduce erosion by runoff because they impart roughness to the soil surface and therefore reduce runoff velocity. Buried geotextiles reduce erosion to a lesser extent as they cannot control rainsplash erosion and actually increase erosion if they are backfilled with loosely consolidated, and therefore highly erodible, material. Over time, natural fiber geotextiles biodegrade and become less effective as vegetation establishes on the slope while artificial fiber mats continue to perform.

These measures also control soil erosion by contributing nutrients, organic matter, and carbon to the soil, thus encouraging microbial activity and improving soil structure and aggregate stability, which in turn further reduces the potential for soil erosion. Vegetation can also be used to reduce slope length and steepness (and thus erosion risk) when grown in strips parallel to the contour. The strips may be of different crops (strip cropping) or as a single field strip across the slope. The strips restrict the

potential for runoff generation and encourage deposition of eroded sediment within the strips so that local slope gradients may be reduced. Common in areas where field boundaries (e.g., hedges) have been removed to increase field size and allow agricultural intensification, field grass strips can be established along former field boundaries, reinstating shorter slope lengths and reducing erosion risk. As farm machinery can traverse the field strips, this simple but effective soil erosion control measure may be adopted willingly by farmers.

Agroforestry systems, in which perennial trees or shrubs grow alongside annual crops, also help to control soil erosion. The trees or shrubs form across-slope barriers to runoff, thus reducing slope lengths and soil losses on the slope.

Wind Erosion Control. Vegetation is also widely used to control wind erosion. By exerting a drag on air flow, vegetation can reduce the velocity of the wind below that required to initiate erosion and can enhance the deposition of soil already being carried in the air. The most common use of vegetation is in the form of shelter belts spaced at regular intervals at right angles to the direction of the erosive winds. Traditionally, trees have been used to form shelterbelts but they need to be supplemented by shrubs (bushes) and grasses to provide cover close to the ground surface since some 70–75% of the soil eroded by wind is carried within the lower 1 m of the atmosphere. The most important characteristic of a shelterbelt is its density or porosity. If too dense, the belt acts like a solid wall and air flowing over the top will create erosive eddies in the lee. The belt therefore needs to be porous so that some air passes through the belt but at a low velocity. Conversely, if the belt is too sparse, the velocity of the wind is not reduced. A porosity of 40–50% is generally considered optimum. Belts are usually designed to maintain wind speeds below 75% of the speeds recorded in open ground. To achieve this, they need to be spaced at intervals of between 12 and 17 times the belt height.

While many shelterbelts comprise single rows of trees, some consist of three rows made up of alternating nurse trees, durable trees, and bushes. The advantage of these mixed belts is they can withstand the loss of an individual species due to disease or pest since other species within the belt can fill any gaps. They also provide a habitat for wildlife.

In arable farming, shelterbelts are considered boundary features and are placed around field edges. However, the principles of shelterbelts can also be used to design in-field shelter systems in which crops are alternated on a strip system, aligned at right angles to the wind. Thus, strips of barley, spaced at 5–10 m intervals, can be alternated with strips of carrots, onions, or sugar beet to control erosion. The barley is sown in late autumn to allow sufficient time for it to emerge before the main crop is drilled in the spring. The barley strips reduce wind velocity until the cover of the main crop is sufficient to protect the soil, at which point the barley is killed off with a selective herbicide so that it does not compete with the crop.

Since shelterbelts require a certain amount of time to grow and become effective, they are not appropriate

where immediate control of wind erosion is required. In these situations, windbreaks made from brushwood or plastic netting can be used. The design principles remain the same, aiming for 40–50% porosity and spacings of 12–17 times the windbreak height. Brushwood fences are commonly constructed on coastal dunes to reduce wind velocities and stabilize the mobile sand, which can then be seeded or planted with shrubs and trees to provide long-term stability.

Soil Management Methods of Soil Conservation

The inherent susceptibility of a soil to erosion depends on the soil type and this is difficult to adjust. However, soil management can affect other factors that can increase or decrease soil erosion. For example, intensive arable production can lead to severe soil degradation as soil is broken down by numerous, invasive tillage practices into a loose, fine tilth seedbed. The resulting small soil aggregates are highly susceptible to dispersion, detachment, and transport by rainfall, runoff, and wind. Soil structural stability is further reduced by loss of carbon through oxidation as tillage exposes deeper soil layers to the air, and by loss of soil faunal biomass destroyed during soil disturbance.

The way in which soil is managed in arable production systems therefore has important impacts on soil erosion control. Farming operations carried out up and down the slope will increase the risk of runoff generation while contour cultivations will reduce these risks significantly. Contour ridging enhances this principle by creating raised ridges across slope to reduce local slope length, reduce effective catchment area for runoff generation, and intercept any runoff that is generated. A development of contour ridging is the use of “tied” ridges, where the ridges are joined at regular intervals by “ties” running up and down slope, so creating small basins that trap runoff and eroded soil.

Reducing the number and intensity of cultivations may preserve soil structural stability and aggregation. Less disturbed aggregates are able to resist the dispersive forces of rainfall, runoff, and wind as cohesive bonds within the aggregates are maintained. Soil microorganisms and macroinvertebrates, which also help bind soil particles together and increase aggregate stability, are also less disturbed. Fewer, less invasive cultivations mean more of the vegetative biomass remains on or within the soil (in the form of standing stubble, cut stems, and roots) and is available to absorb the rainfall and runoff energy otherwise available for soil detachment and transport.

The relative reduction in soil disturbance and retention of vegetative material on and within the soil are the principles behind the use of conservation tillage for the control of soil erosion. This umbrella term incorporates a continuum of soil management methods such as minimum tillage, zero tillage, mulch tillage, direct drilling, noninversion tillage, subsurface mulch tillage, and plough/plant cultivation. Conservation tillage aims at maintaining soil fertility and controlling erosion but has an added aim of reducing input costs to agriculture, as fewer, less invasive cultivations expend less energy and fuel costs.

Manipulation of the soil surface condition by tillage is used to control local wind speeds at the ground surface in an attempt to control soil loss by wind erosion. On light, friable soils such as sands and silts, primary cultivations (ploughing and pressing) are performed in one direction, followed by drilling at 90° to this. The resultant tilth is rough and cloddy and imparts maximum friction to air flow, thus reducing wind speed and the ability to detach soil. Tillage techniques can also be used to minimize water erosion, for example, the Aqueel minimizes erosion by maintaining water within surface depressions on the soil surface (8).

Soil conditioners applied to soils susceptible to erosion can perform the dual role of enhancing soil fertility and controlling erosion. Products with high organic matter (such as seaweed extract and tree bark) add nutrients and organic matter to inherent soil fertility, so improving aggregation, structure, porosity, infiltration capacity, and water holding capacity. Soil conditioners reduce soil erosion by either increasing runoff infiltration or increasing resistance to soil detachment. Hydrophilic soil conditioners include bark, limestone, palm oil effluent, and seaweed extract, which increase aggregation and infiltration capacity, so runoff is less likely to occur. The increase in organic matter from these products also helps control erosion by increasing cohesive bonds in the soil, thus improving the stability of aggregates against soil erosion processes. Hydrophobic soil conditioners, including latex, emulsions, and molasses, form a water-repellent film on the soil surface and increase soil resistance to detachment and transport by rainsplash and runoff.

CONCLUSION

The accelerated loss of soil through water and wind erosion processes is a globally important phenomenon with significant and wide-ranging implications for land and water quality. Controlling the impacts of soil erosion is therefore imperative. A range of control measures are presented, illustrating that soil erosion control can be effectively addressed from a mechanical, agronomic, or soil conservation perspective, depending on the environment, skills, time, and resources available.

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WATER TABLE CONTRIBUTION TO CROP EVAPOTRANSPIRATION

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Evapotranspiration (ET) is the transfer of water vapor from the soil root zone to the atmosphere via the process of photosynthesis. Water vapor can also move directly from the ground surface to the atmosphere. The rate of ET depends on several climatic, soil water, and crop factors, including rainfall, relative humidity, wind speed, air temperature, solar radiation, cloud cover, soil moisture, and stage of crop growth. The plant transpires at the potential rate if soil water is not limiting, which depends on the soil water content.

Water is held in the soil root zone in three distinct phases (see Fig. 1): hygroscopic water, capillary water, and gravitational water. Because of the high tension of the hygroscopic water, it is not available to plants. Both capillary water and gravitational water are available for crop uptake in the ET process.

As the soil water near the soil surface evaporates, the water content near the surface is reduced and the moisture tension increases. Consequently, the tension gradient exceeds the gravitational gradient, and moisture will move upward from the water table toward the soil surface. If root zone soil moisture is between field capacity

and saturation, the crop will transpire at the potential rate. In this case, gravitational water contributes to the ET. However, once the soil moisture drops below field capacity, ET is met from the capillary water. In shallow water table soils, the water table contributes to part of the ET, which is known as capillary rise or upward flux. Depending on the position of the water table, the soil type, and the potential ET, part or all of the crop ET may be met through upward flux. If all ET is not met by upward flux, the remainder is taken from the capillary water. As the gravitational water is depleted, and if there is no supply of water to the soil profile either by precipitation or irrigation, the water table will continue to drop. The water table can be further lowered by drainage and seepage. In arid regions, if irrigation is limited, crops depend on the upward flux of groundwater to meet ET demands. However, in regions of deep groundwater, the upward flux is very small, lower than crop ET, and the plants begin to wilt. They thus exhibit signs of drought stress.

The basic capillary rise equation ($h = 2\gamma \cos \alpha / \rho g R$) governs the height of the capillary rise (h) and is a function of the pore size distribution of the soil particles. Gardner (1) proposed the following equation to estimate the steady upward flux from a static water table:

$$q_{\max} \approx Aad^{-n} \quad (1)$$

where $q_{\max}$ is the maximum evapotranspiration rate; A , a , and n are constants (n is a function of soil texture and is higher for coarser textured soils); and d is the depth to the water table.

Equation 1 shows that the ET rate decreases with water table height and will decrease more strongly for coarse textured soils. When the water table is close to the soil surface (low values of d), the maximum ET will be limited by the prevailing climatic conditions. At lower water table depths (high values of d), the ET will be limited by the transmissivity of the soil. Obreza and Pitts (2) developed relationships between upward flux and water table depth for four locations in Florida (Fig. 2).

The ability to meet crop ET from shallow water tables is used in a process known as subsurface irrigation (subirrigation). By making use of perforated plastic pipes installed below the soil surface (primarily for drainage), drainage outflows can either be restricted or water can be pumped back into the plastic pipes during the growing season, thus maintaining a shallow water table. Upward flux from the water table helps to meet the crop ET demands, which is a high energy and water efficient method of irrigation. Water application efficiency can be as high as 95%. It is also a low-cost irrigation system, because there is no need for additional investments in aboveground hardware. The only additional cost is a structure on the tile drain outlet.

The height at which the water table ought to be maintained depends on the crop. Shallow-rooted vegetable crops will need a water table of 30–50 cm below the soil surface. For deeper rooted crops, e.g., corn and soybeans, a water table of 60–75 cm will meet the crop ET requirement. Close monitoring of the water table depth is advantageous, especially in the wetter humid

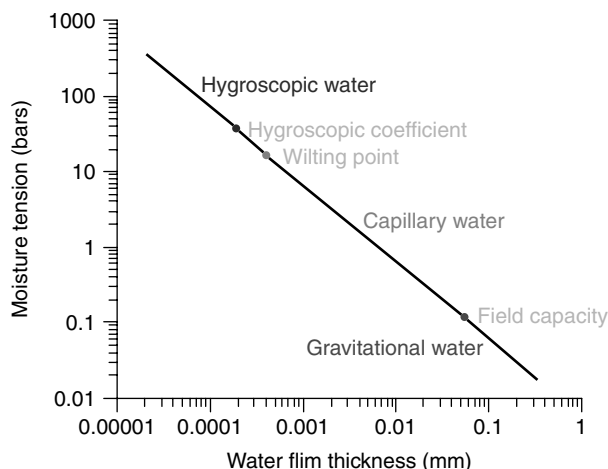


Figure 1. Soil water potential. (Copyright © 1999–2004 Michael Pidwirny).

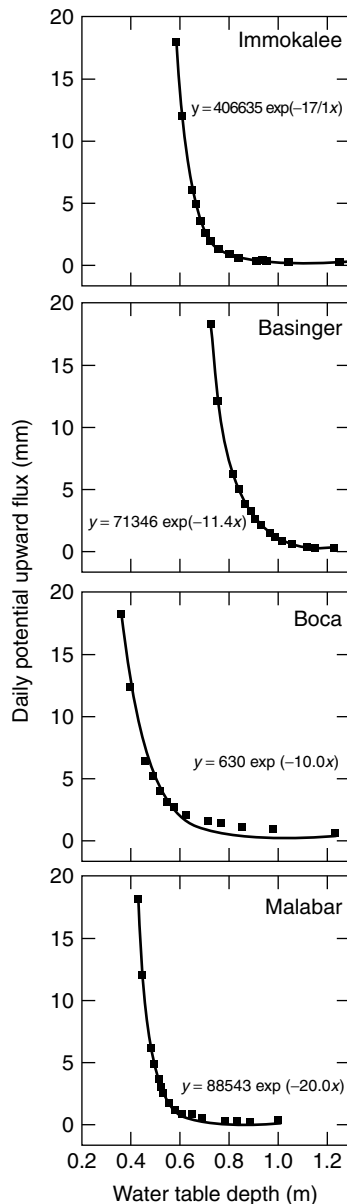


Figure 2. Upward flux vs. water table depth for 4 locations in Florida (2).

regions. Should heavy rainfalls occur, the water table may rise into the crop root zone and cause water logging. In this case, the tile outlet should be opened to lower the water table. The spacing between the tile lines also influences the height of the water table. Narrow-spaced pipe laterals will ensure a more uniform water table distribution. In just the same way that soil type influences upward flux, so too does it influence the lateral pipe spacing. Therefore, the ET rate and the soil type must be taken into account when designing a subirrigation system.

Soils affected by salinization and water logging pose a severe management problem. Salts tend to move upward by capillary action and enter the root zone, thus restricting crop growth and leading to an accumulation of salt in the root zone. Under these conditions, upward groundwater

movement is harmful and subsurface drainage is required to lower the water table.

In forested regions, and in tree plantations, there could be large amounts of ET by the tree canopy, thus lowering the water table. This phenomenon is used in some parts of the world affected by salinity and water logging. Salt-tolerant trees and shrubs are planted to control water logging and salinity, which is known as biodrainage.

Modern irrigation practices are being promoted to make better use of the soil as a reservoir to store precipitation and irrigation water. By encouraging field practices that enhance infiltration and improving soil tilth, there is less surface runoff, and more water is stored at shallow depths that can be extracted by the plants' roots to meet the ET demands. This process is also known as rainwater harvesting in rainfed agriculture. The overall effect is that water is conserved in the soil rather than in large open reservoirs, thus reducing irrigation infrastructure costs, operation and maintenance costs, and water distribution losses. Higher irrigation efficiencies are thus obtained.

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CROP EVAPOTRANSPIRATION

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Crop evapotranspiration is a physical process in which water passes from the liquid to the gaseous state while moving from the soil to the atmosphere. It refers to both evaporation from soil and vegetative surfaces and transpiration from plants. These two processes are considered together because they occur simultaneously in nature, and it is difficult to separate them when measured or estimated.

Crop evapotranspiration refers to the water losses from a cropped field, and this term is frequently used in irrigation scheduling and irrigation water management.

Crop evapotranspiration is affected by weather factors (radiation, air temperature, humidity, and wind speed), crop parameters (crop type, variety, and development stage), and local environmental and management conditions (soil salinity, application of fertilizers, tillage practices, planting density, presence of diseases and pests, windbreaks, irrigation practices, presence of groundwater and impermeable soil horizons, etc.). Accordingly, it is possible to distinguish between crop evapotranspiration

under standard (optimal) conditions and crop evapotranspiration under nonstandard conditions. Standard conditions refer to disease-free, well-fertilized crops, grown in large fields, under optimal water supply, and achieving full production for the given climatic conditions. Nonstandard conditions are defined as those that differ from standard conditions. *Crop evapotranspiration under standard conditions* is usually denoted as ET_c , and *crop evapotranspiration under nonstandard conditions* is marked as ET_{c_adj} or ET_{c_corr} , where the suffixes “adj” and “corr” refer to “adjusted” and “corrected,” respectively. Standard conditions concur with the optimal conditions of well-managed irrigation fields, so, crop evapotranspiration under nonstandard conditions is always lower than or equal to crop evapotranspiration under standard conditions. Therefore,

$$ET_{c_adj} = K_s ET_c \quad (1)$$

where K_s is the reduction coefficient ($K_s < 1$) that accounts for the difference between standard and nonstandard conditions.

MEASUREMENTS OF CROP EVAPOTRANSPIRATION

Crop evapotranspiration can be determined by measurements and estimates. In general, measurements are carried out at agrometeorological stations and experimental sites to evaluate estimating procedures and to calibrate and derive parameters of estimation methods. Different types of lysimeters and methods based on a water and energy balance are used to measure crop evapotranspiration.

Lysimeters are tanks located in the fields and filled with soil of the same type as the surroundings. In lysimeters and their surroundings of at least 100 m around (to allow the establishment of an equilibrium sublayer), crops are grown under homogeneous management practices (standard or nonstandard) to measure the amount of water lost by evapotranspiration. The lysimeters provide direct measurements of crop evapotranspiration and can be grouped in three categories: weighing, where evapotranspiration is measured by weighing the changes in soil water content in the tank; nonweighing of a constant water table where the water table level is maintained at the same level inside as outside the lysimeter, and nonweighing of the percolation type, where water stored in the soil is determined by sampling or neutron methods and rainfall and percolation are measured. Weighing lysimeters give the most accurate estimates for short time periods (e.g., 1 hour), and they are recommended for arid and semiarid areas; nonweighing lysimeters of the constant water table type are particularly suggested for areas where a shallow groundwater table exists, nonweighing lysimeters of the percolation type are frequently used in areas of high precipitation.

The *water balance method* is a direct measurement technique based on the principles of conservation of mass which state that the difference in flow into and out of a controlled volume of soil during a specified time interval corresponds to the changes in soil moisture. This method requires monitoring many parameters such

as irrigation water supply, precipitation, surface and subsurface lateral flow, and deep percolation. The accuracy of crop evapotranspiration estimates by this method is usually low due to difficulties in measuring and controlling one or more terms of the water balance equation.

Energy balance methods are based on the energy balance equation which states that the net flux density of radiation is balanced by losses of latent heat, sensible heat, and by soil heat flux. The Bowen ratio approach is the commonly used energy balance method. It relies on the flux gradient theory and expresses the latent heat flux of evapotranspiration through the measured values of net radiation, ground heat flux, the difference in air temperature between two heights above the canopy, and the corresponding difference in vapor pressure between the same two heights. This method can be used for crop evapotranspiration estimates in hourly and shorter time intervals.

ESTIMATES OF CROP EVAPOTRANSPIRATION

The estimate of crop evapotranspiration relies on the so-called two-step approach where in the first step, a reference evapotranspiration is determined and then, in the second, the crop evapotranspiration (ET_c) is calculated as a product of reference evapotranspiration (ET_o) and crop coefficient K_c :

$$ET_c = K_c ET_o \quad (2)$$

In Eq. 2, the reference evapotranspiration term refers primarily to the climatic demand, and the crop coefficient accounts mainly for the specific crop characteristics and partially for the management practices (e.g., frequency of soil wetness, etc.).

The concepts of reference surface and reference evapotranspiration (ET_o) estimates have been evaluated significantly during the second half of the twentieth century when they have passed from free water surface and empirical and semi empirical formulas to a hypothetical crop of fixed characteristics and more mechanistic equations.

Numerous equations for estimating reference crop evapotranspiration have been developed and successively modified to account for specific climatic conditions and availability of input information. Some of them in widespread use are presented in Table 1 along with the required input weather data and recommended timescale of applications.

In general, the methods for ET_o estimates do not have global validity and require local validation and calibration when applied in different environments. The accuracy of an ET_o estimate has a strong relation with the number and quality of required input information, local climatic conditions, timescale of applications, etc. For example, the Hargreaves method (1) could be the most appropriate for monthly ET_o estimates, provided that only temperature data are available. Nevertheless, it should not be used for daily ET_o estimates, especially when days are hot and dry and wind speed is relevant. The same can be stated for the radiation method which shows good results in humid climates and tends to underestimate ET_o in

Table 1. Some Methods for Reference Evapotranspiration Estimates, Weather Variables Needed, and Suggested Timescale of Application^a

Method	Temperature	Humidity	Wind Speed	Sunshine or Radiation	Evaporation	Timescale			
						Hour	Day	Week	Month
Blaney–Criddle	+	–	–	*	–				X
Hargreaves	+	–	–	*	–				X
Pan evaporation	–	–	–	–	+			X	X
Radiation	+	–	–	+	–			X	X
Penman	+	+	+	+	–		X	X	X
Penman–Monteith	+	+	+	+	–	X	X	X	X

^a + must be measured; – is not necessary; * estimate required; X recommended timescale of application

arid climates. The pan evaporation method integrates the effects of different weather variables on evaporation from a specific free water surface and requires calibration to account for the local environment. None of these methods should be used for daily ET_0 estimates due to limited input information and approximations introduced in the equations. For ET_0 estimates on daily and shorter timescales, more mechanistic methods are needed that rely on the full set of input weather variables and combine both aerodynamic and energy balance principles.

The first physically based combination equation was derived by Howard L. Penman in 1948 to describe the evaporation from a free water surface, and until now, this equation has remained the basis of numerous successive modifications and approaches. The main drawback of the Penman approach (2,3) is in the assumption that the evaporation process originates on the outer surface of the leaves rather than within their substomatal cavities. This lack of generality was satisfactorily overcome by John L. Monteith (4,5), who combined energy balance, aerodynamic, and surface parameters in an evapotranspiration equation formally valid for uniform vegetation of any type, fully covering the ground, and in any state of water status.

During the last decades of the twentieth century, many tests around the world have proved the robustness and consistency of the Penman–Monteith approach (6–9) and suggested it as the standard method for reference evapotranspiration estimates. Consequently, it was recommended to introduce a new concept of reference evapotranspiration surface necessary to standardize computational procedures for estimating reference evapotranspiration. These procedures should be used in different regions and climates of the world and thus, avoid ambiguities when the values of reference evapotranspiration have to be estimated and compared.

New reference surfaces and corresponding methodologies for evapotranspiration estimates represent the results of investigation by two groups of scientists:

- the FAO (Food and Agricultural Organization) Expert Group on the Revision of FAO Methodologies for Crop Water Requirements, which published the FAO Irrigation and Drainage paper 56 and also made it available on the Internet at www.fao.org/docrep/X0490E/x0490e00.htm, and
- the ASCE-EWRI (American Society of Civil Engineers—Environmental Water Resources Institute)

Task Committee on Standardization of Reference Evapotranspiration, which released the first draft of Technical Report in July 2002.

The FAO Expert Group defined the reference surface as “a hypothetical reference crop with an assumed crop height of 0.12 m, a fixed average surface resistance of 70 s/m (over 24 hours), and an albedo (reflectance of evaporating surface) of 0.23”.

The ASCE-EWRI Task Committee distinguishes between two reference surfaces: the first is a short crop with an approximate height of 0.12 m, (similar to clipped grass and corresponding to most of the reference locations in the world), and the second is a tall crop with an approximate height of 0.50 m (similar to full-cover alfalfa and resembling many reference locations across the United States).

Then, provided that a standard set of measured input data is available (including air temperature and humidity, wind speed and incoming solar radiation), the reference evapotranspiration (ET_0) can be calculated by the standardized form of the Penman–Monteith equation:

$$ET_0 = \frac{0.408\Delta(R_n - G) + \gamma \frac{C_n}{T + 273} U_2 (e_s - e_a)}{\Delta + \gamma(1 + C_d U_2)} \quad (3)$$

where ET_0 is the reference evapotranspiration (mm day^{-1} for a daily time step or mm hour^{-1} for an hourly time step), R_n is the net radiation, ($\text{MJ m}^{-2} \text{day}^{-1}$ for a daily time step or $\text{MJ m}^{-2} \text{hour}^{-1}$ for an hourly time step), G is the soil heat flux density ($\text{MJ m}^{-2} \text{day}^{-1}$ for a daily time step or $\text{MJ m}^{-2} \text{hour}^{-1}$ for an hourly time step), T is the mean daily or hourly air temperature at 1.5 to 2.5 m height ($^{\circ}\text{C}$), Δ is the slope of the saturation vapor pressure vs. temperature curve ($\text{kPa } ^{\circ}\text{C}^{-1}$), γ is the psychrometric constant ($=0.066 \text{ kPa } ^{\circ}\text{C}^{-1}$), e_s is the saturated vapor pressure at air temperature (kPa), e_a is the prevailing actual vapor pressure (kPa), U_2 is the mean wind speed measured at 2 m height (m s^{-1}), and C_n and C_d are numerator and denominator constants, respectively, that change with the type of reference surface and calculation time step.

Equation 3 represents the simplest way to estimate the reference evapotranspiration from both reference surfaces (short and tall vegetation) and for different (hourly and daily) time steps. The values for C_n and C_d are derived by simplifying several terms within the Penman–Monteith

Table 2. The Values for C_n and C_d to be Used in Equation 3 (after Ref. 11)

Calculation Time Step	Short Reference Surface (Grass-Like)		Tall Reference Surface (Alfalfa-Like)	
	C_n	C_d	C_n	C_d
Daily	900	0.34	1600	0.38
Hourly during daytime	37	0.24	66	0.25
Hourly during nighttime	37	0.96	66	1.7

equation. They are presented for different reference surfaces and various time steps in Table 2.

The calculation procedures for the reference evapotranspiration and intermediate meteorological parameters ($R_n, G, e_s, e_a, \Delta, \gamma$, etc.) are reported in many books and publications: ASCE Manual 70 (6), FAO 56 (10), the ASCE-EWRI Task Committee report (11), etc. Moreover, free software (REF-ET) for standardized calculations of reference evapotranspiration and intermediate parameters is available at www.kimberly.uidaho.edu/ref-et. An example of a reference evapotranspiration estimate using the standardized form of the Penman–Monteith equation is given here for both grass-like and alfalfa-like surfaces.

Example of a Calculation daily (for a location in southern Italy on June 10, 2002)

Given:

Net radiation $R_n = 14 \text{ MJ m}^{-2} \text{ day}^{-1}$; soil heat flux $G = 0$ (can be neglected especially for daily ET_o estimates); minimum air temperature $T_{\min} = 14^\circ\text{C}$; maximum air temperature $T_{\max} = 28^\circ\text{C}$; wind speed at 2 m height $U_2 = 1.2 \text{ m s}^{-1}$; minimum relative humidity (at $T_{\max}$) $RH_{\min} = 50\%$; maximum relative humidity (at $T_{\min}$) $RH_{\max} = 80\%$.

Calculation. The slope of the saturation vapor pressure curve Δ is calculated for mean air temperature $T = 21^\circ\text{C}$ as

$$\Delta = \frac{4098 \left[0.6108 \exp \left(\frac{17.27T}{T + 237.3} \right) \right]}{(T + 237.3)^2}$$

$$= \frac{4098 \left[0.6108 \exp \left(\frac{17.27 * 21}{21 + 237.3} \right) \right]}{(21 + 237.3)^2}$$

$$= 0.1528 \text{ kPa}/^\circ\text{C}$$

The saturation vapor pressure is calculated as the average value of the saturated vapor pressures at $T_{\min}$ and $T_{\max}$:

$$e_{s_T_{\min}} = 0.6108 \exp \left[\frac{17.28T_{\min}}{T_{\min} + 237.3} \right]$$

$$= 0.6108 \exp \left[\frac{17.28 * 14}{14 + 237.3} \right] = 1.60 \text{ kPa}$$

$$e_{s_T_{\max}} = 0.6108 \exp \left[\frac{17.28T_{\max}}{T_{\max} + 237.3} \right]$$

$$= 0.6108 \exp \left[\frac{17.28 * 28}{28 + 237.3} \right] = 3.78 \text{ kPa}$$

$$e_s = \frac{e_{s_T_{\min}} + e_{s_T_{\max}}}{2}$$

$$= \frac{1.60 + 3.78}{2} = 2.69 \text{ kPa}$$

The actual vapor pressure is calculated as the average value of the vapor pressures at $T_{\min}$ and $T_{\max}$:

$$e_{a_T_{\min}} = \frac{RH_{\max}}{100} e_{s_T_{\min}} = \frac{80}{100} 1.60 = 1.28 \text{ kPa}$$

$$e_{a_T_{\max}} = \frac{RH_{\min}}{100} e_{s_T_{\max}} = \frac{50}{100} 3.78 = 1.89 \text{ kPa}$$

$$e_a = \frac{e_{a_T_{\min}} + e_{a_T_{\max}}}{2} = \frac{1.28 + 1.89}{2} = 1.58 \text{ kPa}$$

Then, for a short (grass-like) reference surface and daily ET_o estimate, the constants C_n and C_d can be obtained from Table 2 ($C_n = 900$ and $C_d = 0.34$), and the reference evapotranspiration can be estimated from Equation 3 as

$$ET_o = \frac{0.408\Delta(R_n - G) + \gamma \frac{C_n}{T + 273} U_2 (e_s - e_a)}{\Delta + \gamma(1 + C_d U_2)}$$

$$= \frac{0.408 * 0.1528 * (14 - 0) + 0.066 * \frac{900}{21 + 273} * 1.2 * (2.69 - 1.58)}{0.1528 + 0.066 * (1 + 0.34 * 1.2)}$$

$$= 4.65 \text{ mm/day}$$

Similarly, assuming that the input data remain the same, for a tall (alfalfa-like) reference surface, the constants C_n and C_d can be obtained from Table 2 ($C_n = 1600$ and $C_d = 0.38$), and ET_o can be estimated as

$$ET_o = \frac{0.408\Delta(R_n - G) + \gamma \frac{C_n}{T + 273} U_2 (e_s - e_a)}{\Delta + \gamma(1 + C_d U_2)}$$

$$= \frac{0.408 * 0.1528 * (14 - 0) + 0.066 * \frac{1600}{21 + 273} * 1.2 * (2.69 - 1.58)}{0.1528 + 0.066 * (1 + 0.38 * 1.2)}$$

$$= 5.43 \text{ mm/day}$$

The example shows that the reference evapotranspiration is greater from a tall alfalfa-like surface than from a short grass-like surface. This is due to the fact that both the aerodynamic and surface resistance are lower for the tall alfalfa-like surface than for the short grass-like surface.

The estimate of the reference evapotranspiration on an hourly basis can be done by using the same equation and appropriate values for C_n and C_d from Table 2. Moreover, the net radiation R_n should be expressed in $\text{MJ m}^{-2} \text{ hour}^{-1}$, and the soil heat flux G should be taken into consideration as a portion of the energy term.

The Hargreaves equation represents an alternative solution for the estimate of reference evapotranspiration when input data are limited only to the values of minimum and maximum air temperature (the data widely available and published in hydrometeorological bulletins worldwide). This simple empirical equation has shown good performance when applied on a monthly scale and also on shorter timescales (e.g., weekly), when the wind speed is not strong and the climate is humid rather than dry.

Besides the minimum and maximum air temperatures ($T_{\min}$ and $T_{\max}$, respectively), the Hargreaves method (1) employs in the estimate the extraterrestrial radiation (R_a) which is a function of the latitude of the location and the day of the year (Julian day) under consideration that can be easily obtained from agrometeorological tables or calculated by using the following set of formulas:

$$R_a = 37.586 d_r (\omega_s \sin \varphi \sin \delta + \cos \varphi \cos \delta \sin \omega_s) \quad (4)$$

where

d_r : relative distance Earth–Sun (rad)

$$d_r = 1 + 0.033 \cos \left(\frac{2\pi}{365} J \right) \quad (5)$$

J : day of the year (Julian day) number (nondimensional),

$$J = \text{integer} (30.42M - 15.23) \quad (6)$$

M : month number (1–12) (nondimensional),

ω_s : sunset hour angle (rad),

$$\omega_s = \arccos(-\tan \varphi \tan \delta) \quad (7)$$

φ : latitude of location (rad),

δ : solar declination (rad),

$$\delta = 0.4093 \sin \left(\frac{2\pi}{365} J - 1.405 \right) \quad (8)$$

Then, the widely accepted form of the Hargreaves equation for an ET_o estimate (mm day^{-1}) for a reference grass surface is

$$ET = 0.0023 \frac{R_a}{\lambda} (T + 17.8) (T_{\max} - T_{\min})^{0.5} \quad (9)$$

where R_a is the extraterrestrial radiation ($\text{MJ m}^{-2} \text{ day}^{-1}$), λ is the latent heat of vaporization (MJ kg^{-1})

$$\lambda = 2.501 - (2.361 \times 10^{-3}) T \quad (10)$$

T is the average temperature ($^{\circ}\text{C}$)

$$T = \frac{T_{\min} + T_{\max}}{2} \quad (11)$$

and multiplier (0.0023), exponent (0.5), and the value of 17.8 in the second multiplier of the equation are the

parameters which should be validated and eventually calibrated for different locations.

Example of a Calculation monthly (for a location in southern Italy in June 2002)

Given:

Minimum and maximum air temperature $T_{\min} = 14^{\circ}\text{C}$ and $T_{\max} = 28^{\circ}\text{C}$; latitude of location is $\varphi = 40^{\circ} = 40 \times 3.14/180 = 0.697778 \text{ rad}$; the month is June which means $M = 6$.

Calculation. Using the following sequence of Equations (6–5–8–7–4), the extraterrestrial radiation ($\text{MJ m}^{-2} \text{ day}^{-1}$) is calculated as

$$J = \text{integer} (30.42M - 15.23)$$

$$= \text{integer} (30.42 \times 6 - 15.23) = 167$$

$$d_r = 1 + 0.033 \cos \left(\frac{2\pi}{365} J \right)$$

$$= 1 + 0.033 \cos \left(\frac{2 \times 3.14}{365} 167 \right) = 0.96818$$

$$\delta = 0.4093 \sin \left(\frac{2\pi}{365} J - 1.405 \right)$$

$$= 0.4093 \sin \left(\frac{2 \times 3.14}{365} 167 - 1.405 \right) = 0.40715256$$

$$\omega_s = \arccos(-\tan \varphi \tan \delta)$$

$$= \arccos[(-\tan 0.697778)(\tan 0.40715256)]$$

$$= 0.935331$$

$$R_a = 37.586 d_r (\omega_s \sin \varphi \sin \delta + \cos \varphi \cos \delta \sin \omega_s)$$

$$R_a = 37.586 \times 0.96818 [0.935331 (\sin 0.697778)$$

$$\times (\sin 0.40715256) + (\cos 0.697778)(\cos 0.40715256)$$

$$\times (\sin 0.935331)] = 29.267$$

Then, the average temperature and the latent heat of vaporization are calculated as

$$T = \frac{T_{\min} + T_{\max}}{2} = \frac{14 + 28}{2} = 21^{\circ}\text{C}$$

$$\lambda = 2.501 - (2.361 \times 10^{-3}) T = 2.501 - 0.049581$$

$$= 2.45142 \text{ MJ kg}^{-1}$$

Finally, the grass reference crop evapotranspiration is

$$ET = 0.0023 \frac{R_a}{\lambda} (T + 17.8) (T_{\max} - T_{\min})^{0.5}$$

$$ET = 0.0023 \frac{29.267}{2.45142} (21 + 17.8) (28 - 14)^{0.5}$$

$$= 3.99 \text{ mm/day}$$

This means that the grass reference evapotranspiration on a monthly basis for June 2002 was $3.99 \times 30 = 119.7 \text{ mm month}^{-1}$.

The average daily value of the ET_o estimate using the Hargreaves method (3.99 mm day^{-1}) is significantly

lower than the value (4.65 mm day^{-1}) obtained from the estimate using the Penman–Monteith equation. This is due to the fact that the impact of wind speed and humidity are not taken into consideration in the Hargreaves equation. However, this equation can be used on a monthly basis to estimate an average reference evapotranspiration. Moreover, the estimates of ET_0 using the Hargreaves equation can be improved by adapting the parameters to the local conditions.

The second step in the crop evapotranspiration estimate is the determination of crop coefficient K_c and the calculation of ET_c by Equation (2). The crop coefficient is a nondimensional factor that represents the ratio between the crop and reference evapotranspiration. The crop coefficient integrates the features of a typical field crop that change during its growing season and differ from the characteristics of a reference surface, which has a constant appearance and is a complete ground cover. These features comprise the crop height and crop aerodynamic properties (roughness), albedo (reflectance), canopy resistance, and soil evaporation from a cropped field.

K_c values vary from crop to crop during the growing season and depend also on weather conditions and irrigation practices (Fig. 1). It is possible to distinguish between the single crop coefficient K_c and the dual crop coefficient where the K_c is split into the basal crop coefficient (K_{cb}) accounting for crop transpiration and the evaporation coefficient (K_e) referring to soil evaporation. The use of a single K_c approach is recommended in regular applications for management purposes and the development of irrigation scheduling; the dual K_c approach should be applied only for specific purposes (e.g., research) and on a daily basis.

K_c values for different crops are given in many technical materials (FAO 24, FAO 33, FAO 56, etc.) referring to short grass as the reference surface. Nevertheless, it is important to emphasize that most K_c values reported in the literature are obtained with the modified Penman equation (FAO 24) and they should be revised and updated when used with the FAO Penman–Monteith method or

some other equation for a reference evapotranspiration estimate. In fact, when crop evapotranspiration has to be calculated, the K_c values should be applied in conjunction with the equations with which they were obtained.

The planting months and the lengths of crop development stages for some widely distributed species are given in Table 3, and an indicative list of K_c values for different crop growth stages is presented in Table 4. The K_c values are given, assuming that the reference evapotranspiration is calculated for a grass-like surface. Both tables are prepared on the basis of the FAO data and some other investigations. However, these data should be used only in the absence of more reliable values.

Example of an estimate of evapotranspiration for a tomato crop growing in the Mediterranean region with the planting date of April 15th.

Calculation Procedure. The lengths of the growing stages are taken from Table 3 and K_c values from Table 4. Then, the growing cycle of tomato can be presented graphically as in Fig. 2.

The K_c values can be determined daily by using a simple linear interpolation. For example, on June 10th, the K_c value is 0.92. Then, assuming that the reference evapotranspiration is 4.65 mm day^{-1} (as calculated in the example here before for the grass-like reference surface using the Penman–Monteith equation), the tomato evapotranspiration for June 10th is estimated by using Eq. 2 as

$$ET_c = K_c ET_0 = 0.92 * 4.65 = 4.28 \text{ mm/day}$$

Finally, if the tomato crop is not growing under standard (optimal) conditions, the crop evapotranspiration can be adjusted to the real conditions by applying Eq. 1. The values of reduction coefficient K_s are always lower than one and depend on the specific site conditions to which the crop is exposed during its growing cycle. In any case, in real management, checking the results of ET_c

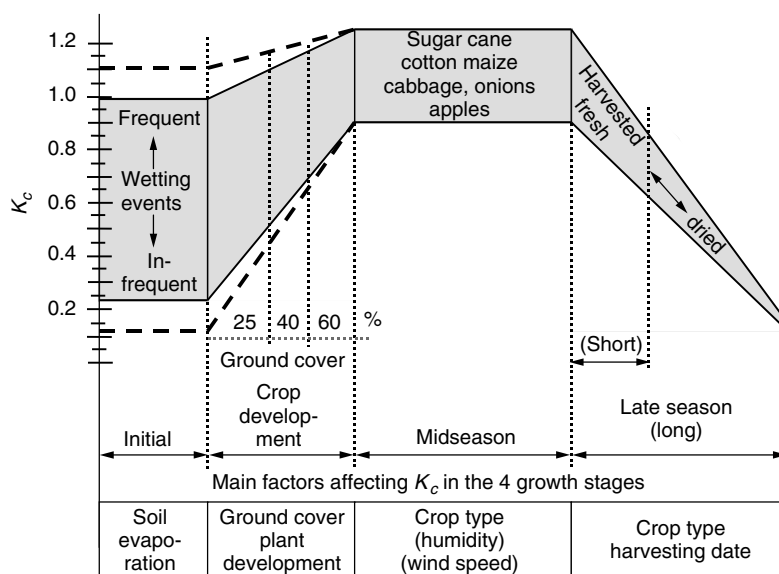


Figure 1. Main factors affecting the crop coefficient (K_c) during the four growing stages (adapted from FAO 56).

Table 3. Planting Months and Lengths of Crop Development Stages for Some Important Crops in Different Regions of the World (Adapted from FAO 56)

Crop	Planting Month	Length of Crop Development Stages					Region
		Initial	Develop.	Mid	Late	Total	
Tomato	January	30	40	40	25	135	Arid regions
	Apr/May	35	40	50	30	155	Calif., USA
	January	25	40	60	30	155	Calif. desert, USA
	Oct/Nov	35	45	70	30	180	Arid regions
	April/May	30	40	45	30	145	Mediterranean
Potato	Jan/Nov	25	30	30/45	30	115/130	Semiarid climate
	May	25	30	45	30	130	Continental climate
	April	30	35	50	30	145	Europe
	Apr/May	45	30	70	20	165	Idaho, USA
	December	30	35	50	25	140	Calif. desert, USA
Sugar beet	March	30	45	90	15	180	Calif., USA
	June	25	30	90	10	155	Calif., USA
	September	25	65	100	65	255	Calif. desert, USA
	April	50	40	50	40	180	Idaho, USA
	May	25	35	50	50	160	Mediterranean
	November	45	75	80	30	230	Mediterranean
	November	35	60	70	40	205	Arid regions
Soybeans	Dec	15	15	40	15	85	Tropics
	May	20	30/35	60	25	140	Central USA
	June	20	25	75	30	150	Japan
Cotton	March–May	30	50	60	55	195	Egypt, Pakistan, Calif.
	March	45	90	45	45	225	Calif. desert, USA
	September	30	50	60	55	195	Yemen
	April	30	50	55	45	180	Texas
Sunflower	April/May	25	35	45	25	130	Medit., Calif.
Barley/Oats/Wheat	November	15	25	50	30	120	Central India
	March/April	20	25	60	30	135	35–45° Lat.
	July	15	30	65	40	150	East Africa
	April	40	30	40	20	130	
	November	40	60	60	40	200	
	December	20	50	60	30	160	Calif. desert
Winter Wheat	December	20	60	70	30	180	Calif., USA
	November	30	140	40	30	240	Mediterranean
	October	160	75	75	25	335	Idaho, USA
Maize (grain)	April	30	50	60	40	180	East Africa (alt.)
	Dec/Jan	25	40	45	30	140	Arid climate
	June	20	35	40	30	125	Nigeria (humid)
	October	20	35	40	30	125	India (dry, cool)
	April	30	40	50	30	150	Spain, Calif.
	April	30	40	50	50	170	Idaho, USA
Sorghum	May/June	20	35	40	30	130	USA, Pakis., Medit.
	March/April	20	35	45	30	140	Arid regions
Rice	Dec; May	30	30	60	30	150	Tropics; Medit.
	May	30	30	80	40	180	Tropics
Grapes	April	20	40	120	60	240	Low latitudes
	March	20	50	75	60	205	Calif., USA
	May	20	50	90	20	180	High latitudes
	April	30	60	40	80	210	Midlatitudes (wine)
Citrus	January	60	90	120	95	365	Mediterranean
Apples/Pears	March	20	70	90	30	210	High latitudes
	March	20	70	120	60	270	Low latitudes
	March	30	50	130	30	240	Calif., USA
Olives	March	30	90	60	90	270	Mediterranean

Table 4. Indicative Crop Coefficient Values for Different Growing Stages of Some Important Crops under Optimal Management Conditions and Reference ET Calculated for a Grass-like Surface (Adapted mainly from the FAO 56)

Crop	$K_{c_initial}^a$	K_{c_mid}	K_{c_end}	Maximum Crop Height (m)
Tomato	0.4–0.6	1.15–1.20	0.70–0.90	0.60
Potato	0.4–0.6	1.15	0.60–0.70	0.40
Sugar beet	0.4–0.5	1.20	0.70	0.50
Soybean	0.3–0.5	1.15	0.50	0.5–1.0
Cotton	0.4–0.5	1.15–1.20	0.60–0.70	1.2–1.5
Sunflower	0.3–0.5	1.0–1.15	0.40	2.0
Barley	0.3–0.5	1.15	0.25	1.0
Oats	0.3–0.5	1.15	0.25	1.0
Spring Wheat	0.3–0.5	1.15	0.25–0.40 ^b	1.0
Winter Wheat				
• with frozen soils	0.4	1.15	0.25–0.40 ^b	1
• with nonfrozen soils	0.7	1.15	0.25–0.40 ^b	1
Maize, Field	0.3–0.5	1.20	0.35–0.60 ^c	2.0
Sorghum				
• grain	0.3–0.5	1.0–1.10	0.55	1–2
• sweet	0.3–0.5	1.20	1.05	2–4
Rice	1.05	1.20	0.90	1.0
Grapes				
• table or raisin	0.30	0.85	0.45	2.0
• wine	0.30	0.70	0.45	1.5–2.0
Citrus, no ground cover				
• 70% canopy	0.70	0.65	0.70	4
• 50% canopy	0.65	0.60	0.65	3
• 20% canopy	0.50	0.45	0.55	2
Citrus, active ground cover or weeds				
• 70% canopy	0.75	0.70	0.75	4
• 50% canopy	0.80	0.80	0.80	3
• 20% canopy	0.85	0.85	0.85	2
Apples/Pears				
• no ground cover, killing frost	0.45	0.95	0.70	4
• no ground cover, no frost	0.60	0.95	0.75	4
• active ground cover, killing frost	0.50	1.20	0.95	4
• active ground cover, no frost	0.80	1.20	0.85	4
Olives (40 to 60% ground cover)	0.65	0.70	0.70	3–5

^aThe higher values should be adopted when the surface is frequently wetted.

^bThe higher value is for the hand-harvested crop.

^cThe higher value is for harvesting at high grain moisture.

estimates obtained from equations with the results of on-field monitoring of soil and/or plant water status is highly recommended.

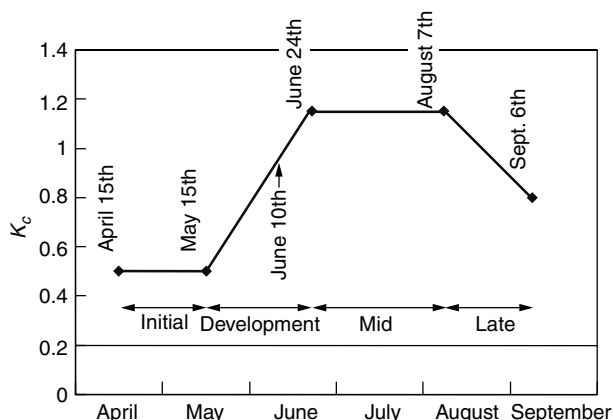


Figure 2. The evaluation of K_c values of a tomato crop during the growing season.

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WATER POLLUTION FROM FISH FARMS

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WASTE OUTPUT

Wastes from aquaculture plants include all materials used in the process that are not removed from the system during harvest. The principle wastes from aquaculture are uneaten feed, excreta, chemicals, and therapeutics. In addition, the term “waste” can also refer to pathogens and dead or escaped fish.

Generally, the quantity of waste is closely connected to the culture system used. *Intensive* farm systems, typically monoculture of carnivorous finfish in the temperate zone reliant on artificial feed, may cause serious local pollution. So-called semi-intensive farm systems are supplied natural feed sources, such as vegetation, oil cakes, cereal bran, and organic-chemical fertilizers. The latter systems dominate the tropical/subtropical production of herbivorous or omnivorous fish, e.g., the major production of carps and tilapia, and the waste output to the surrounding waters is much lower than from intensive fish farms. This is illustrated in Fig. 1, where the nutrient budget is estimated for intensive farms (cage or flow-through raceway systems) and for semi-intensive farms (Nile tilapia produced in static pond water). In highly intensive farms, as much as 74–84% of the nutrient inputs are released into surrounding waters, whereas only 2–6% of the nutrients are released from semi-intensive farms. Semi-intensive production of carps takes place mostly in static ponds only drained during harvest, where supplied nutrients not assimilated in fish are mainly trapped in the pond sediments. The Chinese production of carps and tilapia in freshwater constitutes more than one third of the global aquaculture production (China: 12 million MT

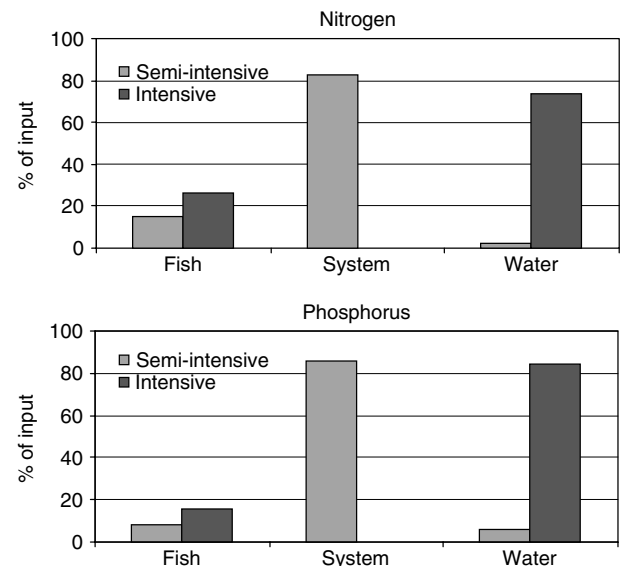


Figure 1. A comparison of the nutrient budget of a semi-intensive and an intensive aquaculture system (see text). After Edwards (2).

in 1997, 1). Thus, the retained fraction of nutrients in the pond sediments is vital to avoid serious water pollution.

In distinct regions, fish farms represent a dominating water pollution source. Along the western and northern coast of Norway, cage farming of salmon and trout contributed about 55% of P and 17% of N of the total input in 1998 (see Fig. 2). Since then, the annual production has increased from c. 400,000 MT to c. 600,000 MT. The total nutrient input to this part of the coastline has strongly increased over the last 20 years because of the growing aquaculture industry. The average specific loading was 10.8 kg P and 52.0 kg N per ton of produced fish. For population equivalents (p.e.): 6000 p.e. as Total P and 4000 p.e. as Total N per MT produced fish. Besides, uneaten feed and feces from cage farms load the recipient with biodegradable organic matter that may lead to significant oxygen depletion [corresponds to 300–500 kg as BOD₇ per MT produced fish, (4)].

A wide range of chemicals are used in the aquaculture industry, including disinfectants, chemotherapeutants (antibacterial, antifungal, and antiparasitic compounds), pigments incorporated into feeds, and compounds employed in and applied to construction materials (e.g., antifoulants) (5). Of particular interest is the use of chemotherapeutants because of the quantities discharged into the aquatic environment. In Norway, the only country that has kept records, around 50 MT of antibacterials were introduced into the coastal environment by the aquaculture industry in 1987. The use has, however, fallen to below 1 MT annually because of development of efficient vaccines against bacterial diseases. In semi-intensive, tropical freshwater farms, antibacterials are sparsely applied.

ENVIRONMENTAL IMPACTS

Most reported studies of environmental effects are related to intensive aquaculture. The quality of wastes from fish farms and its temporal variability are important

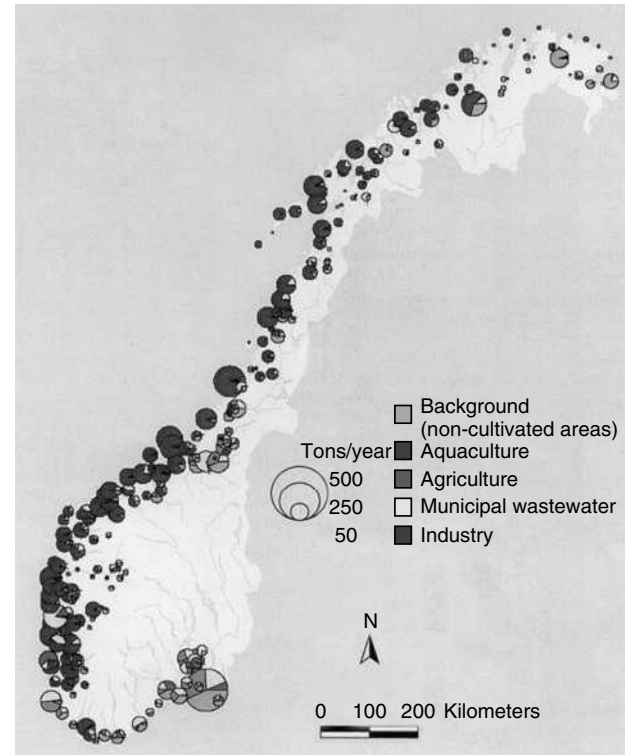


Figure 2. Estimated phosphorus inputs to the Norwegian coastal waters from the main sources in 1998 (3).

determinants of environmental impacts (6). Aquaculture wastes are rich in P with a TP/TN ratio of around 5. Thus, fish farms have been expected to be a potential source for eutrophication in freshwater sources as P is the limiting nutrient. Generally, the eutrophication rates in lakes and reservoirs influenced by fish farming seem to be lower than those predicted by lake models. Håkanson et al. (7) observed very little, if any, increased concentrations of TP in two Swedish lakes influenced by cage farm emissions. The main causes seemed to be linked to (1) direct uptake of feces and waste feed from farmed fish by wild fish, and (2) a significant amount of P from the cages that are eliminated from the lake water by different processes, e.g., sedimentation.

Nutrients emitted from cage farms are found to be highly biologically available for marine algae (8): The bioavailability of P in effluent from aquaculture, agriculture, and municipal sewage was 50/60%, 30%, and 65/70%, respectively, whereas more than 80% of N in all effluent types was available for algal growth. Despite that aquaculture is the dominating nutrient source along the western-northern coast of Norway, the cage farms are located in a zone with huge water exchange (The Gulf Stream) and large recipient capacity.

Deposition of solids from cage farms may strongly effect the benthic ecosystems beneath the cages (6). Such materials have shown to increase the extent of anaerobic sediment layers with dramatic consequences for the original biological communities. Under intensive cage farms, the bacterial activity in anaerobic sediments often causes out-gassing of methane and hydrogen sulphide. In addition, sedimentary phosphorus might be released at oxygen depletion in deeper water. Studies of benthic communities all indicate that detrimental impacts are limited to the immediate vicinity of the intensive cage (or mussel) operation.

Among other consequences of modern aquaculture are environmental effects caused by antibiotic use, e.g., drug residues in water, sediments and organisms, and introduction of exotic species for aquaculture purposes. Welcomme (9) reported that 98 species of fish had been introduced internationally involving inland waters. Also, accidentally or deliberately released farmed fish can affect their local wild forms by behavior and genetic impacts.

ENVIRONMENTAL REGULATIONS

A wide range of regulations and standards control fish farming and the discharge of effluent in parts of the world, such as North America and Europe. Aquaculture relies on precious water resources, and; as populations increase, access to good quality water and the ability to discharge effluents will become more restrictive. In many countries, aquaculture legislation is poorly developed. During the last few years, however, interest has grown to develop a comprehensive regulatory framework for aquaculture whose goals should be to protect the industry,

Table 1. Governmental Regulations for Fish Farm Effluents in Some European Countries per 1999 (11)

Criteria	Country									
	B	DK	F	GR	IRL	NL	N	S	SF	UK
Limitation of production		§			§		§	§	§	
Effluent treatment		§	§	§	§	§			§	§
Limitation of N and P load	§	§	§	§	§		§		§	§
Limitation of organic load	§	§	§	§	§	§	§			§
Feed composition		§	§					§	§	
Feed conversion ratio		§	§				§		§	
Environmental impact assessment		§	§		§		§	§		§
Rules for usage of chemicals					§		§			§
Monitoring of effluent water		§			§		§			§
Taxes on effluent water and/or feeds	§		§			§				§

B: Belgium, DK: Denmark, F: France, GR: Greece, IRL: Ireland, NL: Holland, N: Norway, S: Sweden, SF: Finland, UK: United Kingdom.

the environment, other resource users, and the consumer, and to clarify rules for resource use (10).

A brief review of regulations controlling discharge of effluent from fish farms in ten European countries is presented in Table 1.

The waste load from fish farms can also be reduced by effluent treatment attempts (see WASTE WATER TREATMENT IN FISH FARMS).

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WORLD'S MAJOR IRRIGATION AREAS

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EXTENT OF IRRIGATION

Irrigation is the artificial watering of land by technical means, such as canals, pumps, sprinklers, or drip systems, to supply moisture to crops. Surface irrigation, such as basin and furrow irrigation, in which water is led to the fields through canals and ditches, has been practiced for over two millennia.

About 20% of the world's agricultural land is irrigated, but some 40% of the food and fiber production worldwide takes place on irrigated land. For cereal production, it is close to 60% (1). FAO expects this share to increase further in the next three decades. Much of this expansion will take place in developing countries, increasing their

irrigated land perhaps by as much as 20% before 2030. This is a net expansion, which implies that agricultural land lost to urban growth and soil salinization will be compensated.

At present, more than 60% of the world's irrigated land is in Asia. Table 1 shows the ten top countries and river basins in terms of irrigated land area. About two thirds of the irrigated land in Asia is devoted to rice and wheat production.

Worldwide, about 70% of water withdrawals is supplied to agriculture. Table 2 shows for the ten top irrigation countries the total water withdrawal (for agriculture, industrial, and domestic use) as a percentage of the renewable water resource in the country and the percentage of this total withdrawal used in agriculture.

Although food production to 2020 is expected to increase faster in developing countries than in developed countries, experts suggest that food production will not keep pace with demand. In 1999/2000, developing countries produced 55% of the world grain production and accounted for 61% of world grain consumption. They also imported 231 million tons of grain, equivalent to 72% of worldwide imports, which suggests that developing countries although playing a major role in irrigated agriculture production still depend on international agricultural trade for their food security.

DIFFERENT METHODS OF IRRIGATION

The three main types of irrigation are canal (or surface) irrigation, sprinkler irrigation, and trickle or drip irrigation. In canal irrigation, water is distributed through a network of canals, consisting of one or more main canals conveying water from the source (e.g., a reservoir or a barrage in a river), distributary (secondary) canals from the outlets in the main canal, tertiary canals (called watercourses in the Indian subcontinent) from the outlets in the distributary canals to the farm gate, and field canals. Canal irrigation systems have lower construction and operating costs, simpler maintenance, and lower skilled labor demands than do pressurized delivery systems. A disadvantage of canal irrigation over piped systems is that the soil must convey and infiltrate the water over the fields. Because soil properties are highly variable, both spatially and temporally, designing water-efficient surface irrigation at the farm level is complicated. Much research has been done on determining the optimum flow rate for a given length of furrow and vice versa. It is now claimed that in furrow irrigation design, it is possible to attain an application efficiency (i.e., the ratio of the average depth of water applied to the root zone, and the average depth of water applied to the field) of about 70% (2). However, the reality in the field is probably different. Application efficiency refers to the average depths of application, but the uniformity of distribution is also important. It expresses the often large differences in the depths of water stored in the root zone along the length of the fields resulting from local differences in soil structure and texture and variable infiltration amounts in low and high spots in the field. Precision land leveling and automatic devices (e.g., for the release of surge flows) have

Table 1. Top Ten Countries and River Basins for Irrigated Land

Country	Area (million ha)	% of World Total	River Basin	Area (million ha)
India	59	21	Yangtze	31
China	54	20	Mekong	23
United States	22	8	Xi jiang	19
Pakistan	18	6	Ganges	11
Iran	8	3	Irriwaddy	10
Mexico	6.5	2	Brahmaputra	9
Indonesia	4.8	2	Yellow river	9
Thailand	4.8	2	Indus	8
Turkey	4.5	1.5	Chao Praya	7.5
Bangladesh	4	1.5	Krishna	6

Source: FAOSTAT On-line Statistical Service (<http://apps.fao.org>) and IWMI, 2002. World Irrigation and Water Statistics 2002, International Water Management Institute, Colombo, Sri Lanka.

Table 2. Water Withdrawal as Percentage of Renewable Water Resource, and its Share Supplied to Irrigation

Country	Percent of Water Resource Withdrawn (2000)	Percent of Total Withdrawal Supplied to Agriculture (1995)
India	26	96
China	19	89
USA	17	41
Pakistan	70	96
Iran	54	95
Mexico	17	85
Indonesia	3	94
Thailand	8	85
Turkey	15	83
Bangladesh	1	96

Source: same as Table 1.

improved attainable application efficiency and uniformity of distribution, while reducing the labor requirements of surface irrigation (3).

Sprinkler irrigation and drip (trickle) irrigation eliminate the need for land leveling and allow more frequent water application without alternating saturation and desiccation of the upper soil layers. But most importantly, these newer methods have improved attainable application efficiencies and, hence, lose less water. Nowadays, commercial crops that were traditionally furrow irrigated are increasingly irrigated by drip irrigation methods. For example, a subsurface drip irrigation system was chosen to replace furrow and sprinkler irrigation systems on a commercial sugarcane estate in Swaziland. A postinvestment evaluation indicated that sucrose content increased by 15% and that 22% less water was used in the drip system than in the sprinkler irrigation system it replaced (4). Similar yield increases and water savings were noted in drip irrigated corn (maize) in a project in arid southeast Anatolia, Turkey (5).

CHANGES IN IRRIGATION OBJECTIVES OVER TIME

In the colonial period, the dominant reason for installing irrigation was to prevent famines during drought years. An additional concern was flood control in the deltas of the big rivers in Asia and Africa. Famine prevention necessitated

an irrigation system that would provide some water to many farmers to assure the main harvest and revenue collection for the colonial power with a minimum of control and regulation. In the Asian subcontinent, this was achieved through a form of timed rotation of a continuous flow to different watercourses, in which a shortfall in supplies is shared equally by all users. The system was not expected to provide enough water for maximum yields.

From the 1950s, irrigation rapidly expanded, especially in Asia. The peak in funding by the international development banks for construction of irrigation occurred in the 1970s and 1980s. Internal rates of return on irrigation projects between 1965 and 1985 were in the range of 12–25% (6). This period coincided with the green revolution in Asia when cereal production increased many fold as a result of the introduction of high yielding varieties of rice and wheat, enhanced fertilizer use, and improved irrigation systems. During this period, irrigation's main objectives were to obtain maximum yields per unit land through high water and fertilizer inputs. Concurrently, extensive research was done to determine and predict water requirements and evapotranspiration rates, i.e., evaporation from the soil and transpiration from the growing crop. This topic was central to the efforts to match irrigation supplies as closely as possible with the evaporative demand of the environment to maximize crop production.

After about 1985, irrigation investments dropped sharply because of lower grain prices (from which the urban population benefited more than the farmers), rising construction costs, and hence lower returns on investments. Growing opposition to further large-scale irrigation projects, such as the Namada in India and the Three Gorges Dam in China, contributed to this reduced interest in irrigation investments. Although investments slowed, irrigation development in Asia nevertheless continued unabated. Experience and research have shown that supportive policies and laws have at least as much effect on irrigation development as does the availability of international investments. Specifically, national priorities, strength in the rule of law, a free market economy, local and user participation in planning and management, as well as encouragement of the private sector have been shown to stimulate irrigation development (7).

Pumped groundwater is increasingly used for irrigation. This increase has been at such an extent that many groundwater resources are now overexploited. Recycled drainage water and wastewater have also been added as possible sources of irrigation water. Generally, some treatment of wastewater has been deemed necessary before it can safely be used in crop production. But many developing countries do not have or enforce regulations covering the use of wastewater, or lack the resources to treat wastewater adequately. From the 1990s, the irrigation strategy focused on the control and integrated management of all water resources for agricultural and nonagricultural uses, which made irrigation managers realize that low irrigation efficiencies on the farm do not imply that water is wasted. If the drainage flow is fed back into the canal and reused downstream, and seepage to the groundwater is recycled by pumping, the relevant parameter is no longer application efficiency at the farm level but at the subsystem or system level, or even at the river basin level. For example, the Nile valley in Egypt with its notoriously inefficient irrigation at the farm level has a basin level application efficiency of about 95% because of its extensive reuse of drainage water (8). Hence very little water, and of no further use because of its low quality, is discharged into the Mediterranean Sea. These examples from Third World countries follow earlier developments of a similar nature in the United States and Australia (e.g., respectively, the Imperial Irrigation District with the Colorado River, and the Murray-Darling Basin).

IRRIGATION PERFORMANCE ASSESSMENT

Performance parameters express the degree to which the objectives of irrigation are met. The chosen parameters are usually limited to those pertaining to performance of the irrigation infrastructure and the system management. Because of its large data requirement, the variability in performance among farmers and for different crops is rarely studied. In one instance, where available information allowed quantification of the way farmers managed their irrigation supplies, it was found that around half the cotton farmers irrigated either excessively or deficiently although the average performance was close to the optimum for maximum yield per unit land (9). Excessive irrigation was also found in the Gezira irrigation system in Sudan. Here the variation was attributed to unattended furrow irrigation, aging infrastructure causing siltation, and weed growth in the canals (10). Large variabilities among farmers obviously indicate a significant potential for improvement.

In protective irrigation systems (implying proportional sharing of a limited resource among the widest number of people), performance is evaluated by measuring the flow rates that reach the tail ends of watercourses. This measurement assesses the equity of distribution of the available water to as many farmers as possible. Head-tail differences in water supplies continue to be the case in many large-scale irrigation systems. Failure to recognize and address equity considerations (i.e., fairness as understood by the farmers) in performance assessment

has been a major cause of problems in operation and maintenance of irrigation system.

When the irrigation objective in the second half of the twentieth century shifted from protective to productive irrigation (involving the right to maximum economic return for a limited number of farmers), the performance parameter of choice became the yield per unit of land. Toward the end of the century, water for all uses became a major concern, and irrigation management came to be seen as one element of integrated water resource management. At the same time, performance assessment of irrigation systems was no longer based only on profitability and revenue collection, but it came to include the social objectives of food security, poverty alleviation, and environmental protection.

Because of increasing pressure on available water resources, irrigation managers worldwide are now compelled to improve the performance of their systems. The list of performance parameters has consequently grown and includes all aspects of maintenance and operation of the infrastructure, output per unit irrigation water and per unit of land, financial aspects (e.g., cost recovery), and environmental performance (e.g., quality and quantity of drainage flows). Benchmarking, using a standard against which performance can be assessed, has now been applied to irrigation management. By comparing with the norm, it attempts to identify deficiencies in a system's infrastructure and organizational structures. Benchmarking thus intends to make continuous improvements, especially in the quality and reliability of the services provided to the farmers. It also intends to stimulate an institutional culture that supports innovation and reliability in service providers (11).

IRRIGATION MANAGEMENT

Three main types of irrigation management can be distinguished: commercial estates producing usually one crop (e.g., sugarcane, cotton, tobacco, pineapples, etc.), canal irrigation systems with many small farmers, and individual farmers managing their source of irrigation water, usually a tubewell or small reservoir. Of these, large-scale canal irrigation systems cover the largest area, especially in Asia, and provide employment to millions of people. Improvements in the management of these systems could have a major beneficial impact on food production. This type of management, usually done by state organizations such as irrigation departments, has therefore received the most attention. Common institutional deficiencies in water resource management are described elsewhere (12).

Along field channels, and often significantly above the field channels, farmers control the actual operation of the system. In that part of the system that is controlled by management, including reservoirs and the main and secondary (or distributary) canals, considerable discrepancy often exists between the nominal rules and procedures incorporated in the design and the actual operation of the system.

Because of widespread dissatisfaction with state management of irrigation systems, financial institutions

who invested in irrigation development have stimulated state agencies since the 1990s to hand over all or part of irrigation management to the private sector. Greater participation of the farmers in system management and user-operated and managed irrigation systems as found in developed countries are examples to be followed in the Third World. Privatization was expected to lower the cost of service provision to the farmers while increasing its quality. Conditions for success of privatization include:

- A legal framework either existing or tailor-made to provide the legal background for enforceable contracts
- Quality control and other regulations
- Active participation of the farmers in the implementation of the changes
- Institutions that control and enforce the rules and regulations
- Rights to water and use of the infrastructure
- Impartial adjudication of disputes
- Sound arrangements about the financial responsibilities regarding maintenance of the irrigation infrastructure (13)

An alternative to complete privatization was introduced in Pakistan where irrigation system management was decentralized and the participation of farmers in system management was promoted. Management at some secondary canal level was handed over to the farmers' organizations. A recent evaluation of one of these sites indicates that the water distribution and allocation has become more equitable as a result of the handover, which is particularly beneficial to tail-end farmers (14). Additional benefits were that illegal tampering of outlets had almost ceased, and that cost recovery had improved significantly. However, experience with irrigation management transfer elsewhere in developing countries has not been uniformly positive. Different performance parameters show improvement in some systems but decline in others. In some systems, an increase in farmers' fees for operation and maintenance and in fee collection has occurred, but nevertheless major maintenance is deferred. The effect of management transfer on agricultural production and farm income is equally uncertain.

In recent years, many countries have gained experience with institutional reform of their water sectors, including Australia, India, Philippines, Malaysia, Turkey, Mexico, Argentina, and Chili. Institutional changes will continue to shift toward user-management and water markets, with a more regulatory role for the state. Although the long-term impact of these various efforts has not yet been assessed, the Australian experience clearly demonstrates that if farmers are to pay for the full cost of irrigation services, the irrigation agencies have to become accountable to them.

SUSTAINABILITY AND FUTURE OF IRRIGATED AGRICULTURE

Irrigated agriculture competes for water with other uses and often contributes to degradation of land and water

resources. The net social benefit of irrigated agriculture is the difference between returns to the farmer and society as a whole, and the cost to society associated with its negative aspects (e.g., the cost of disposal of saline and polluted drainage water). Because no general agreement exists on the real benefits and costs of irrigated agriculture, whether irrigated agriculture is sustainable in the long run has become controversial. Many definitions of sustainability exist. A strict definition is that sustainability is achieved when irrigation and drainage are conducted on-farm and within irrigation districts, in a manner that does not degrade the quality of land, water, and other resources (15). Others define sustainability of irrigated agriculture as the ability to continue extracting net positive returns from water and land for an indefinite period of time (16). In contrast to the first definition, the latter one is not inconsistent with some degree of environmental degradation, which implies that it is not necessarily true for all ecosystems that the optimal rate of degradation is zero.

Linked with this issue of sustainability is the question of whether there is enough irrigable land and water for future needs. FAO studies (17) suggest there is still scope for expanding irrigation to satisfy future needs. Their estimates suggest that in developing countries, only half of the potential irrigated land (about 400 million ha) is in use. Moreover, only 20% of this presently unused potential area will be in production by 2030. This assessment takes into account the limitations imposed by the availability of water. In the developing countries as a whole, only about 7% of renewable water resources were withdrawn (18) for irrigation. And the projection is a 14% increase in water withdrawals for irrigation by 2030. Water availability is considered critical when at least 40% of the renewable resource is used for irrigation. Of the countries listed in Table 2, the situation is critical only in Pakistan and Iran. FAO expects this to be the case for all of South Asia and West Asia/North Africa by 2030. Although a 14% increase in water withdrawn for irrigation over a period of 30 years may not appear to be much, the development of additional resources will be increasingly difficult. For example, monsoonal river discharges are part of the unutilized resource, but they are extremely expensive to capture.

In another study (19), the potential demand (i.e., without water supply constraints) and actual consumption (realized demand given the limitations in supply, and calculated for consumption rather than withdrawal) of irrigation water were assessed. This study expects the potential demand in developing countries to grow by 12%, but the actual consumption should grow less because of water scarcity and improvements in yields per unit water. The difference in growth of the potential and realized demands leads over time to lower values of the proportion of the potential demand realized in actual consumption of irrigation water, as shown in Table 3. The expected improvements in water productivity (kg/m^3) in irrigated cereal production range from about one third in sub-Saharan Africa to two thirds in South Asia.

With respect to the potential for irrigation expansion in the industrial countries, most studies agree that there

Table 3. Proportion of Potential Demand Realized in Actual Consumption

	1995	2025
Asia	0.81	0.76
Latin America	0.83	0.75
Sub-Saharan Africa	0.73	0.72
West Asia/North Africa	0.78	0.74

Source: Rosegrant et al. (19).

is less scope and need for expansion in America and Europe. At present some irrigated land in California goes out of production as the water rights are sold to cities for their domestic water supply. In both the United States and Europe, some rainfed agricultural land is now left idle as there is no need for more production. If the need developed, agricultural productivity could probably be enhanced on rainfed land at a lower cost than would be involved in expanding irrigated land. Moreover, future production increases in developed countries are expected to come from higher yields per unit of land and per unit of water.

It has been argued that water pricing might reduce the demand for irrigation water. However, water used for irrigation is not simply an economic resource but also a social good. A change in the price of water would affect the urban price of food, which in turn could have major political consequences. It would be more appropriate to charge for the water actually consumed than for what is supplied in irrigation, but that is practically impossible. Another practical difficulty with water pricing as an instrument for affecting its use is that for the price to have an effect on water demand, it must be significantly increased, most likely more than politically feasible. A case in point is China, where according to the World Bank, water for agriculture is heavily subsidized. A better option for improving water use efficiency in irrigation appears to be rationing water where the demand exceeds supply (20).

A contentious issue is how to improve water productivity (e.g., kg/m³ or \$/m³) when water resources are scarce and higher water rates are not appropriate. One option is through deficit irrigation: applying less water than the crop would need for maximum production per unit of land. Another option is to design a responsive irrigation system in which all farmers have access to water as and when needed, which is expected to lead to less wastage. The latter, however, is difficult and expensive to realize in large-scale irrigation systems with many fragmented small holdings. Providing farmers in deficit irrigation with less water than they would freely demand is expected to encourage farmers to grow more water-efficient and high-value crops. The downside of deficit irrigation, though, could be the possible salt buildup in the root zone in the absence of sufficient leaching of salts from the soil profile. Model studies have shown that on-demand irrigation is profitable where water is relatively abundant or high-value crops are grown that would benefit from extreme reliability and accuracy in irrigation supplies. Yet, deliberately restricting irrigation deliveries at no less than 80% of the potential full demand, crops can still produce well

without salinization of the root zone. The productivity of water (kg/m³) could then be almost 50% higher with deficit irrigation than when water is supplied to meet full crop demand (21).

The first challenge for the future, then, is to improve crop production on irrigated land by managing water better to achieve optimal production per unit of water, which is more easily realized in developed countries than in developing ones, but because of the population growth, the latter have a greater need for more food production. Present thinking is that more food export from industrial countries is not the solution. Imports have to be paid for in hard currency and staple crops should be grown where the people live. However, this could change as a result of globalization and reductions of tariffs in agricultural trade. Hence, for now and the foreseeable future, irrigated agriculture is essential for worldwide food security. The second challenge thus becomes to enhance irrigated agriculture while minimizing the adverse environmental impacts.

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IRRIGATION IN THE UNITED STATES

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CONTRASTED TRENDS

The total amount of farmland, comprising both cropland and pastureland, is slowly decreasing in the United States at 71.2 million hectares (M ha) (176 million acres) in 1998, but irrigated lands kept increasing from 16.7 Mha in 1974, 19.8 Mha in 1982, 19.9 Mha in 1992, and 25 Mha in 2000. The geography of the increase in irrigated lands is contrasted. The national yearly increase for the period 1974–2000 is 1.57%, but east central and Atlantic states witnessed much stronger growth, well above 4%, on average, whereas Arizona, Montana, Nevada, Texas, Oklahoma, and Wyoming saw their irrigated lands shrink overall (Fig. 1) (1).

But this 26-year period masks a three-period evolution. From 1974 to 1982, irrigated surfaces grew on average by 2.18% yearly; sharp increases occurred in states such as South Carolina (+29%/year), Georgia (+22.7%/year), and Minnesota (+19.1%/year); other western states such as Texas and New Mexico already displayed a downward trend in their irrigated surfaces, and eastern States as well such as New Jersey and West Virginia. From 1982 to 1992, the growth is much more moderate: 0.08% per

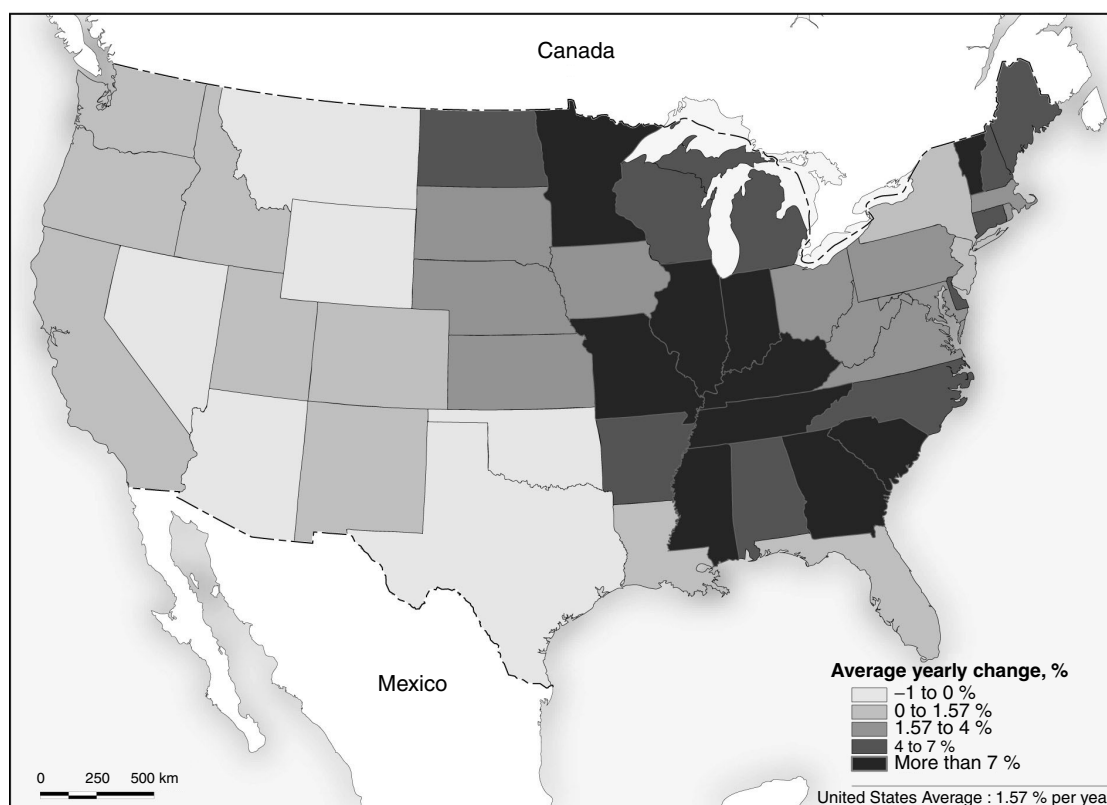


Figure 1. Irrigated surfaces evolution, 1974–2000.

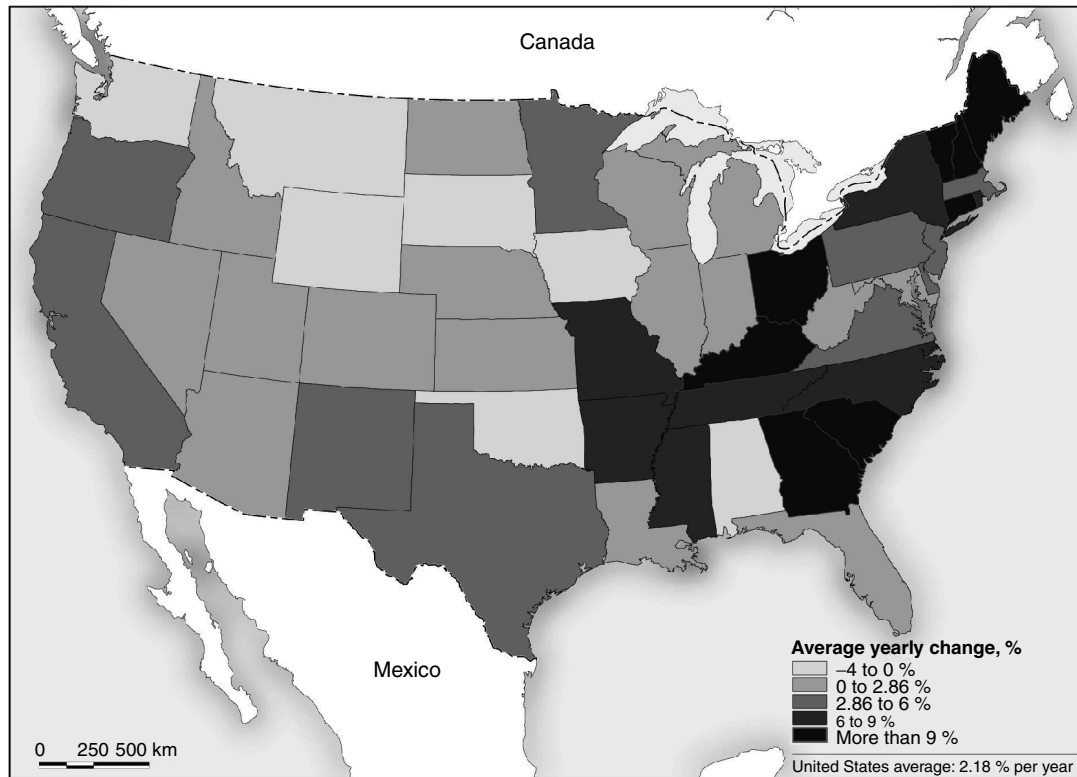


Figure 2. Irrigated surfaces evolution, 1992–2000.

year in the nation, with most states located in the western part of the country witnessed a decrease. Then growth picked up from 1992 to 2000 at a national yearly average of 2.86%. Some western states (Fig. 2) (Oregon, Montana, Wyoming, Oklahoma, South Dakota) still experienced a decline, but others grew appreciably (California: +3.7%/yr; Texas: +3.5%). The strongest growth was experienced in the Atlantic and east central regions, growth rates exceeded 9% in Georgia, South Carolina, Ohio, Kentucky, and several New England states (Tables 1 and 2).

This spectacular growth of irrigated surfaces in the eastern United States can be explained by two factors: a production objective, the desire of farmers to increase yields or switching to more water-demanding crops, such as corn, sugar, vegetables; a financial factor as well; many farmers in this wetter part of the country irrigate their cropland to avoid any risk of yield decrease. Tight markets, strong pressures from banks, and rising debt have made some farmers risk averse.

THE IRRIGATION COUNTRY: WATER IN THE DRY WEST

However strong the irrigation growth might have been in the eastern part of the country and even though a slight decline has begun to take place in a few western states, absolute irrigated surfaces still show that irrigation is concentrated mainly in the western and central United States. In 1974 as well as in 1992 (Figs. 3 and 4), the picture depicted by the maps underlines the dominant share in irrigated surfaces in states west of the 100th meridian: in 1974, irrigated land in California, Idaho,

Colorado, Nebraska, and Texas accounted for 58.3% of all irrigated land. In 2000, irrigated land in the same five states still made up 51% of irrigated land countrywide.

Of course, natural conditions largely explain this spatial pattern: rainfall becomes much scarcer west of the 100th meridian and becomes more abundant again only in the northwestern part in the country, on the Pacific coast (Fig. 5). Most of the west gets less than 20 inches of rain a year; patches of California, Nevada, Arizona, Idaho, Utah, Wyoming, and even Washington receive less than 10 inches. Water is present, however, in the region: aquifers do contain large amounts of water (for instance, the Ogallala aquifer underneath Texas, Oklahoma, and Nebraska). And mountain ranges generate rivers; the Colorado drains a large basin in the area. Tapping this resource became a political objective during the conquest of the West during the second half of the nineteenth century to tame nature and embody the conquest of the land, even though it implied transferring agricultural techniques that belonged to the wetter eastern part of the country. Water schemes were designed to develop the resource, rivers were dammed, and canals built to divert rivers, paid for by the federal budget and delivering water that was and still is largely subsidized. Agriculture is flourishing in the arid west, but it is only thanks to the development of what Donald Worster (3) calls a “hydraulic society,” where water is the main resource and cements the very fabric of the economy and society. This explains why, today, it is still a very political and emotional issue fiercely debated, although the age of large-scale dams and developments seems to be gone.

Table 1. Top 15 States with Increases in Irrigated Surfaces, 1974–2000 and 1992–2000

1974–2000	%/year
South Carolina	11.78
Georgia	10.60
Vermont	9.19
Missouri	8.74
Mississippi	8.72
Indiana	8.05
Minnesota	7.98
Illinois	7.64
Tennessee	7.23
Kentucky	7.20
Maine	6.99
Alabama	6.41
Arkansas	6.18
Michigan	5.76
Delaware	5.59
North Carolina	5.29
1992–2000	%/year
Connecticut	17.20
Maine	17.07
New Hampshire	17.07
South Carolina	11.91
Vermont	11.45
Kentucky	11.44
Georgia	9.87
Ohio	9.51
Missouri	8.18
North Carolina	7.13
New York	7.09
Arkansas	6.61
Rhode Island	6.59
Tennessee	6.34
Mississippi	6.12
New Jersey	5.98

WATER WITHDRAWALS

The geography of water withdrawals for irrigation (Fig. 6) confirms the dominant role of irrigation in agriculture in the West: large withdrawals take place in this part of the country in every state but a few of them and reaches levels as high as 70.1 billion m³/year in California (see Table 3).

But the geography of water withdrawals also shows that eastern states withdraw appreciable volumes of water for irrigation: Florida, of course (9.1 billion m³/year), largely because of the fruit and sugar industries; but also Georgia (2.2 Bm³/year); North and South Carolina (cotton, peanuts, corn, tobacco); New Jersey; Massachusetts (0.33 Bm³/year); and Michigan. In the three latter states, large water withdrawals are largely accounted for by the redevelopment of fruit and vegetable cultures for nearby urban markets. Central States in the Missouri/Mississippi basin also use significant water for irrigation: Nebraska (14.4 Bm³/year), Arkansas (13.3 Bm³/year), Kansas (5.7 Bm³/year), Mississippi (2.1 Bm³/year), Missouri (2.1 Bm³/year), Louisiana (1.8 Bm³/year), and Oklahoma (1.2 Bm³/year).

Some states rely heavily on groundwater for irrigation. In the lower 48 states, 19 states rely on groundwater for

Table 2. Fifteen States Experiencing the Lowest Increase in Irrigated Surfaces, 1974–2000 and 1992–2000

1974–2000	%/year
New Jersey	1.39
Oregon	1.27
Louisiana	1.13
Florida	1.08
Idaho	1.05
California	1.02
Washington	0.70
Colorado	0.65
New Mexico	0.54
Oklahoma	–0.06
Texas	–0.06
Montana	–0.09
Arizona	–0.64
Nevada	–0.70
Wyoming	–0.88
1992–2000	%/year
Illinois	1.33
Wisconsin	0.89
Colorado	0.88
Maryland	0.77
Louisiana	0.57
Indiana	0.46
Arizona	0.26
West Virginia	0.00
Oklahoma	–0.12
Washington	–0.55
South Dakota	–0.58
Montana	–1.73
Alabama	–1.96
Wyoming	–2.88
Iowa	–3.85

more than 40% of irrigation withdrawals; levels are as high as 49.7% for Montana and 49.1% for Kentucky. On the other hand, 17 states pump less than 30% of their irrigation water from aquifers. Reliance on groundwater is not connected to the size of the withdrawals; states along the lower Mississippi and Missouri basins and around the Great Lakes tend to pump a larger proportion of their irrigation water from surface water.

As a result of the huge withdrawals taking place in the West for irrigation, shares of irrigation water in total freshwater withdrawals (Fig. 7) are nearly always above 60%, except for Kansas (56.3%), Oklahoma (40.7%), Texas (34.8%), and North Dakota (12.7%) (Table 4).

In the eastern United States, the share of irrigation is consistently below 15%, except for Arkansas, Florida, Mississippi, Georgia (17.8%) and Missouri (17.4%), and is less than 1% in 12 states (Indiana, Vermont, Kentucky, Iowa, Virginia, New York, Alabama, Ohio, Tennessee, Pennsylvania, West Virginia, and the District of Columbia). Thus, although it is developing fast in the eastern part of the country and may at times in specific places, represent a problem for water supply in the summertime, irrigation remains by far the major demand sector only in the West. This explains the pressure exerted on ground or surface waterbodies by environmental problems such as the overexploitation (Colorado River)

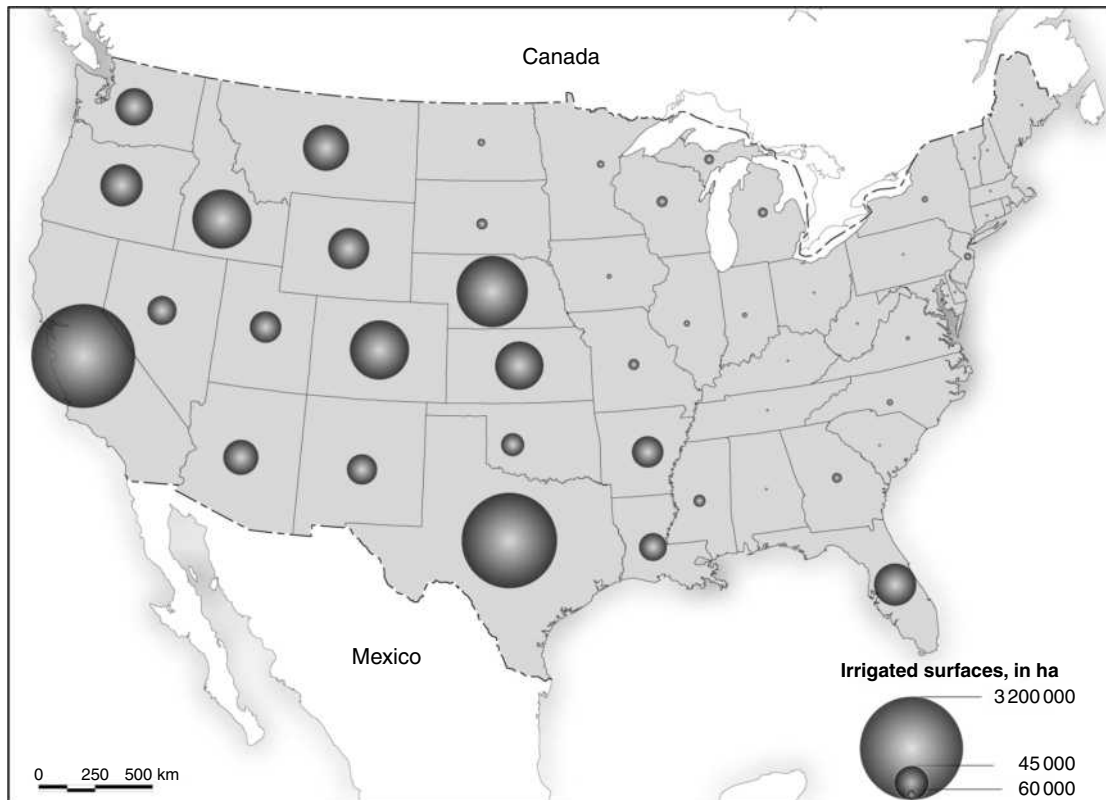


Figure 3. Irrigated surface per state, 1974.

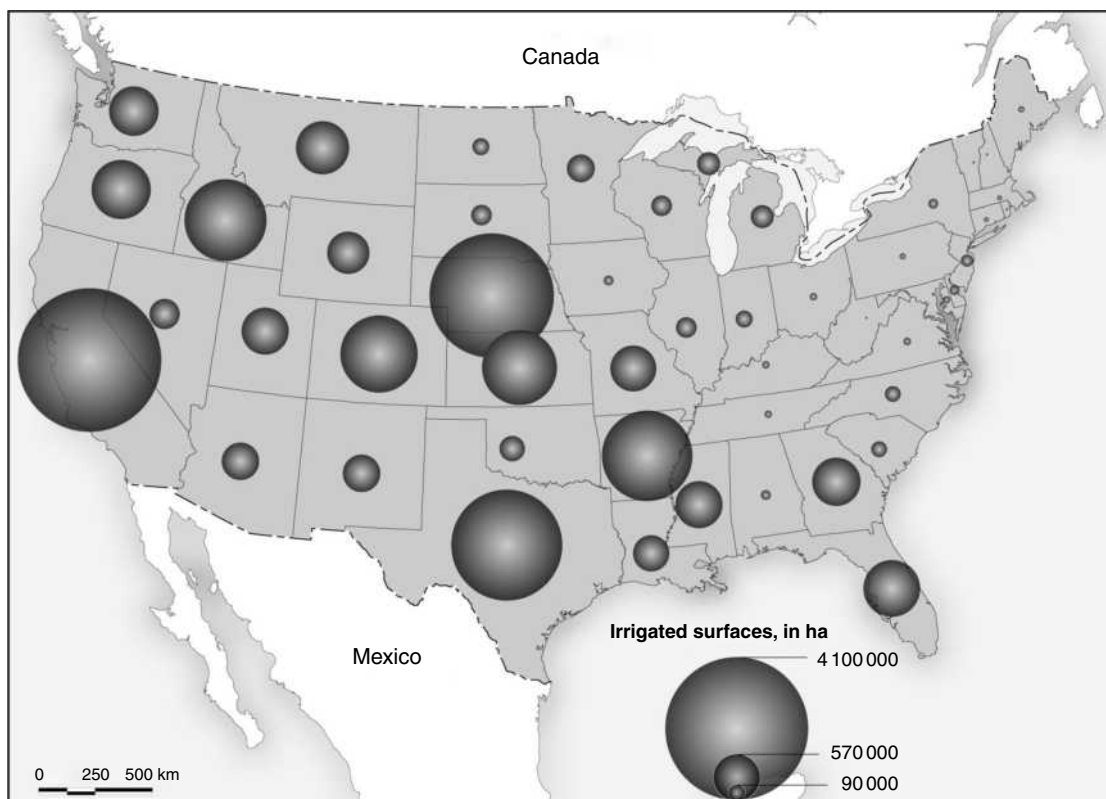


Figure 4. Irrigated surface per state, 2000.

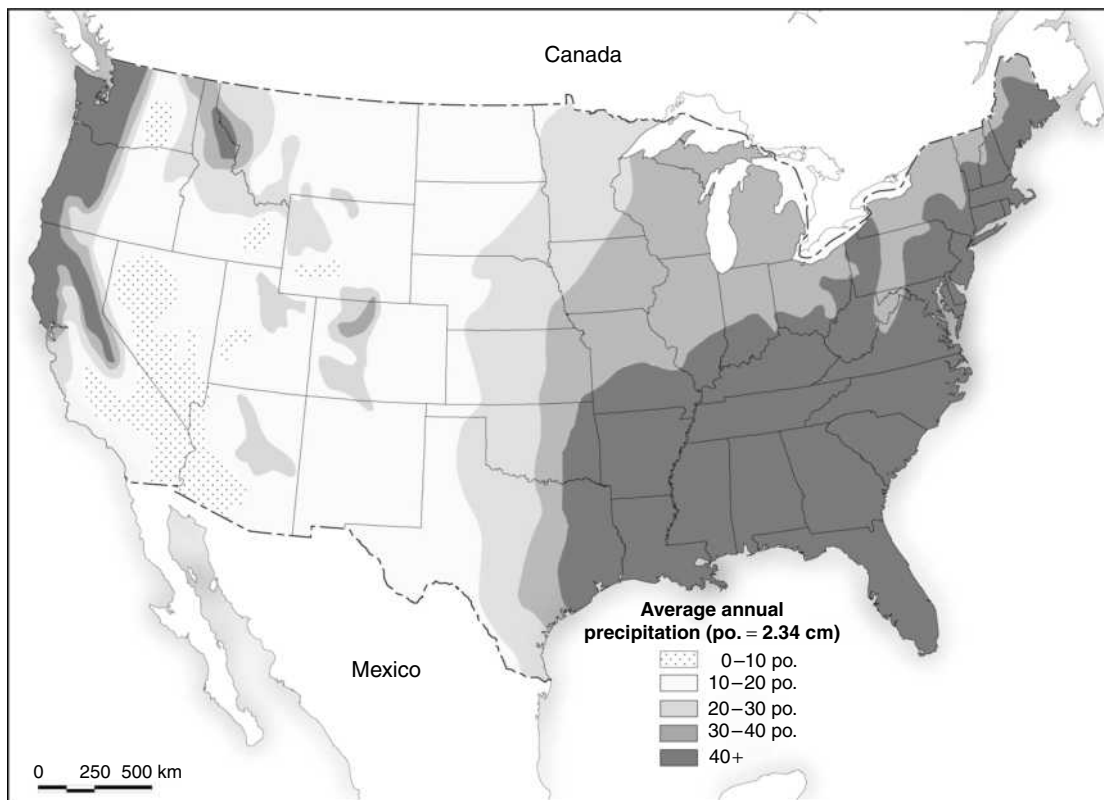


Figure 5. The dry western United States (1,2).

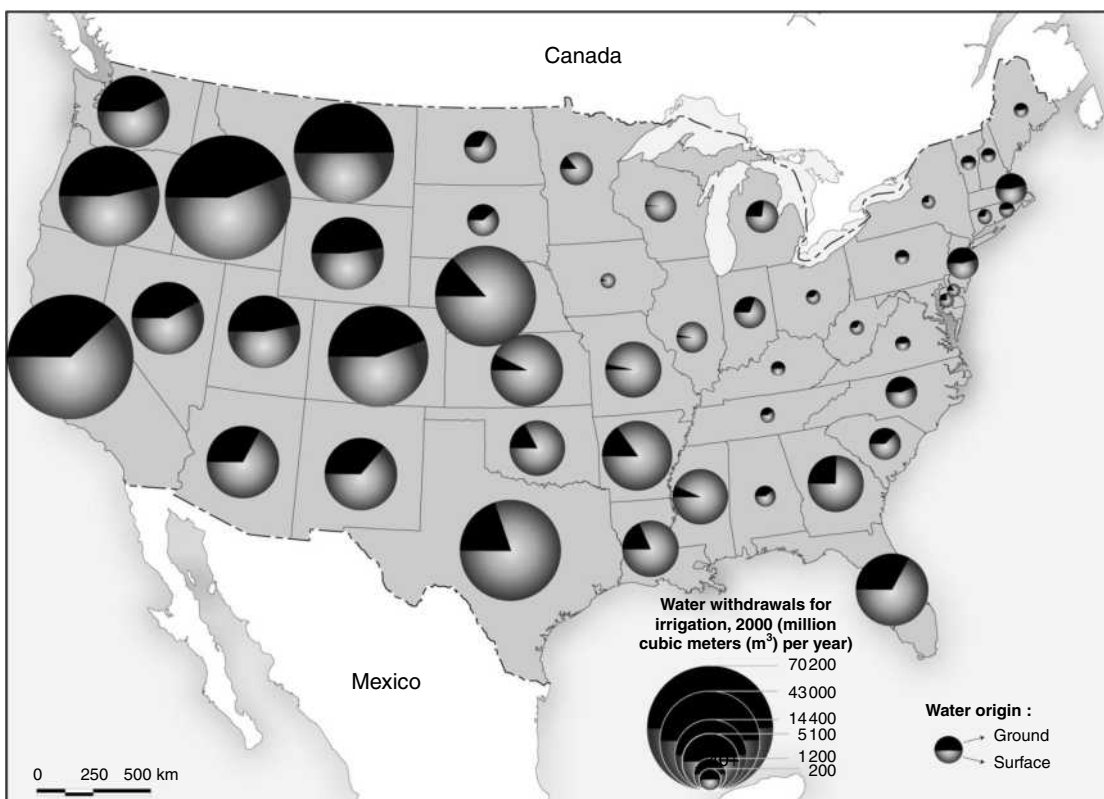


Figure 6. Water withdrawals for irrigation, 2000.

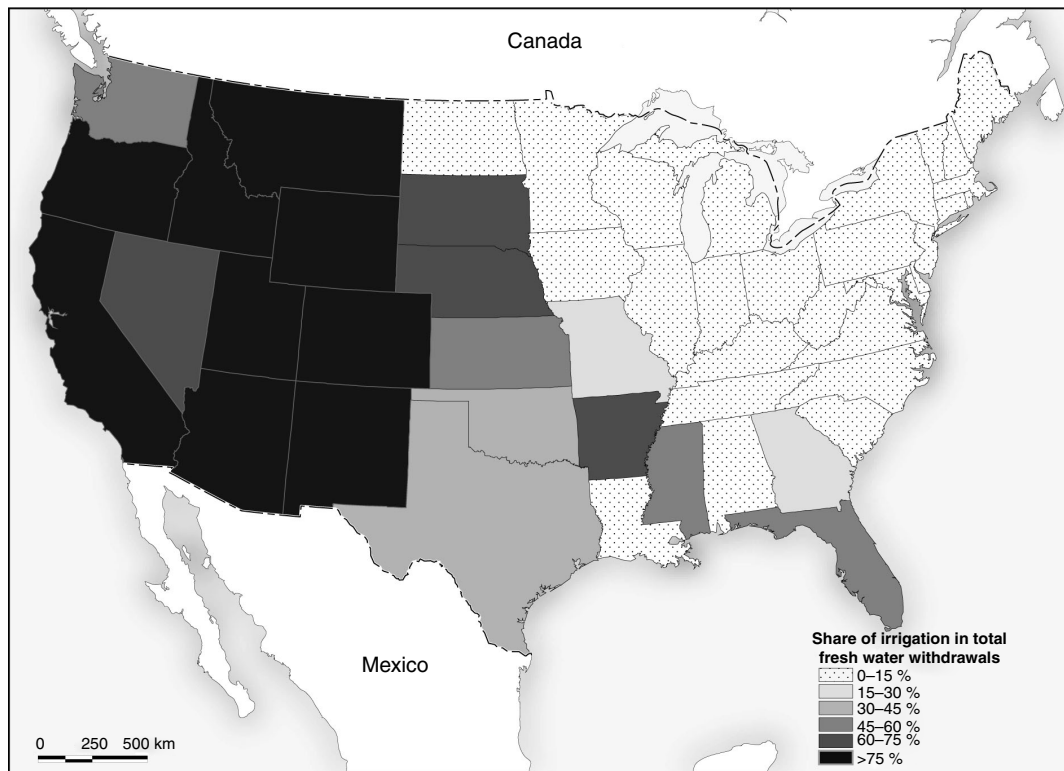


Figure 7. Share of irrigation in total fresh water withdrawals, 2000.

or the salinization of rivers (such as the Gila River). It also entails a developing political debate as to whether the sharing of the resource should leave so much subsidized water to the agriculture sector, as thirsty cities and industries have begun buying up expensive water rights from farmers to satisfy their fast growing needs.

FARMING AND IRRIGATION TECHNIQUES

The agricultural sector displays a contrasting picture, as can be seen in Fig. 8. If absolute water withdrawals for irrigation are very large in the West and if agriculture there remains by far the largest demand sector, the share of irrigated land in total farmland is not necessarily dominant (Table 5).

California, Arkansas, and Mississippi are the only states whose share of irrigated land represents more than half of the total exploited land. The extent of fruit and vegetables crops in California, as well as cotton growing

Table 4. States Whose Share of Irrigation in Total Fresh Water Withdrawals is Greater Than 50%, 2000

State	%
Montana	95.93
Wyoming	91.15
Colorado	90.63
Oregon	87.76
New Mexico	87.73
Idaho	87.28
Utah	81.07
Arizona	80.51
California	79.43
Nevada	74.98
Arkansas	72.66
Nebraska	72.05
South Dakota	70.64
Washington	57.63
Kansas	56.25
Florida	52.70
Mississippi	50.14

Table 3. Top Twenty States for Irrigation Water Withdrawals, 2000 (in Billion Cubic Meters per Year)

California	Idaho	Colorado	Montana	Oregon
70.1	43	29.4	22.5	16.1
Texas	Nebraska	Arkansas	Wyoming	Arizona
15.3	14.4	13.3	12.2	11.5
Utah	Florida	Washington	New Mexico	Kansas
10.3	9.1	7.6	6.4	5.7
Nevada	Georgia	Mississippi	Missouri	Louisiana
5.2	2.2	2.14	2.1	1.8

in the Mississippi valley, accounts for such high shares. Strong fruit and vegetable cropping (Washington, New Jersey, Michigan, Florida, Delaware), sugar (Florida), potato (Idaho), tobacco and peanuts (Georgia) are, in their respective states, dominant cultures where irrigation is extensively practiced, to enable cropping (as for most cultures in the West) or to reduce, even eliminate, the financial risk that could stem from poor rains in an otherwise pluvial agriculture.

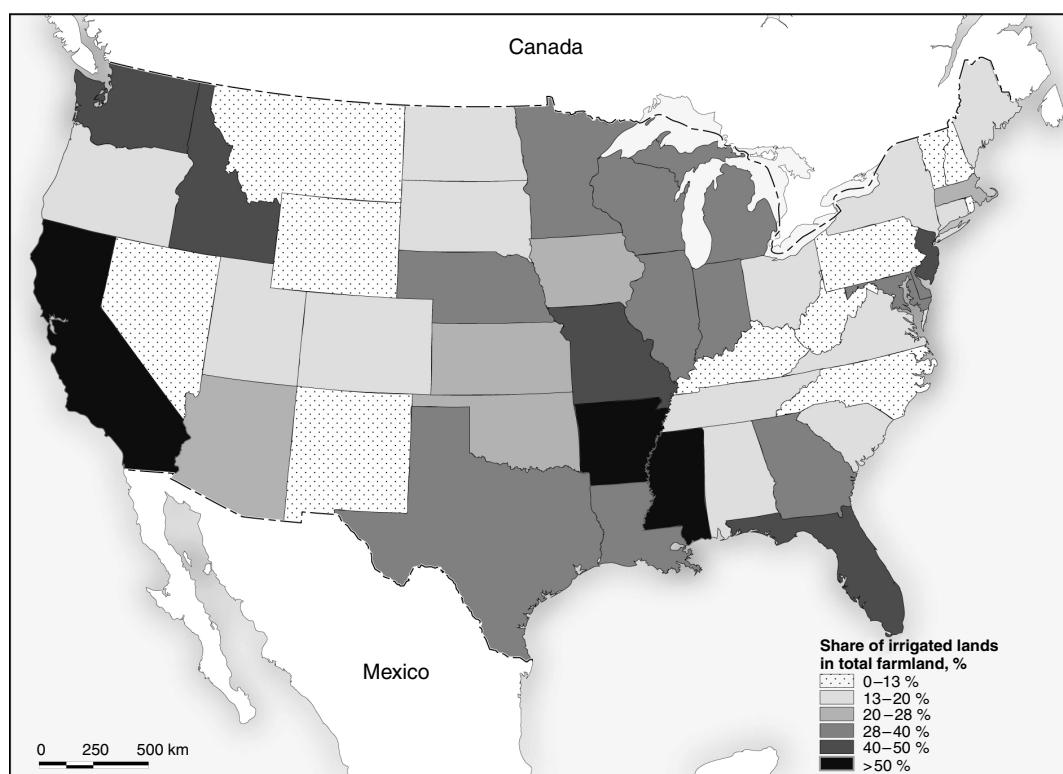


Figure 8. Share of irrigated land in farmland, 1998.

Table 5. States Whose Share of Irrigated Land in Farmland (1998) is Greater Than 30%

State	%
Maryland	31.07
Illinois	31.32
Nebraska	31.37
Georgia	31.89
Delaware	32.22
Minnesota	37.18
Texas	38.16
Louisiana	39.10
Wisconsin	39.26
Michigan	39.78
New Jersey	41.78
Florida	41.78
Missouri	43.02
Idaho	43.33
Washington	45.72
Mississippi	52.36
Arkansas	65.17
California	65.90

These risk-averse farming sectors partly explain why the water application rate per surface unit can be so high in the more humid East (Fig. 9). The low cost of water is certainly a factor, but it rather accounts for profligate rates in the West, as water remains heavily subsidized in this part of the country, whereas farmers in the East, who produce high-value crops such as fruits and vegetables, reason that the extra water spread in their fields is worth the guarantee that they will not suffer from a few percent

loss in their crops because of poor rains (Table 6). This is a consequence of the increasing role of financial institutions in agribusiness, as banks have stepped up their pressure on farmers who, in turn, tend to be willing to spend more on water to avoid potential losses from natural factors.

The cost factor is certainly better reflected in the irrigation technique employed. Irrigation in the East is dominated by the sprinkler technique, an easy-to-implement technology with a higher water productivity than gravity or surface watering. Drip irrigation is also

Table 6. Top Fifteen States for Water Application Rate in Thousand m³/ha, 2000

State	Rate, 000 m ³ /ha
Montana	24.1
Arizona	21.7
Idaho	21.2
Massachusetts	20.97
Wyoming	19.4
Colorado	15.96
Nevada	14.87
Oregon	13.87
Utah	13.54
California	12.8
New Mexico	11.8
Washington	8.9
Florida	8.2
North Carolina	6.8
South Carolina	6
Arkansas	5.4

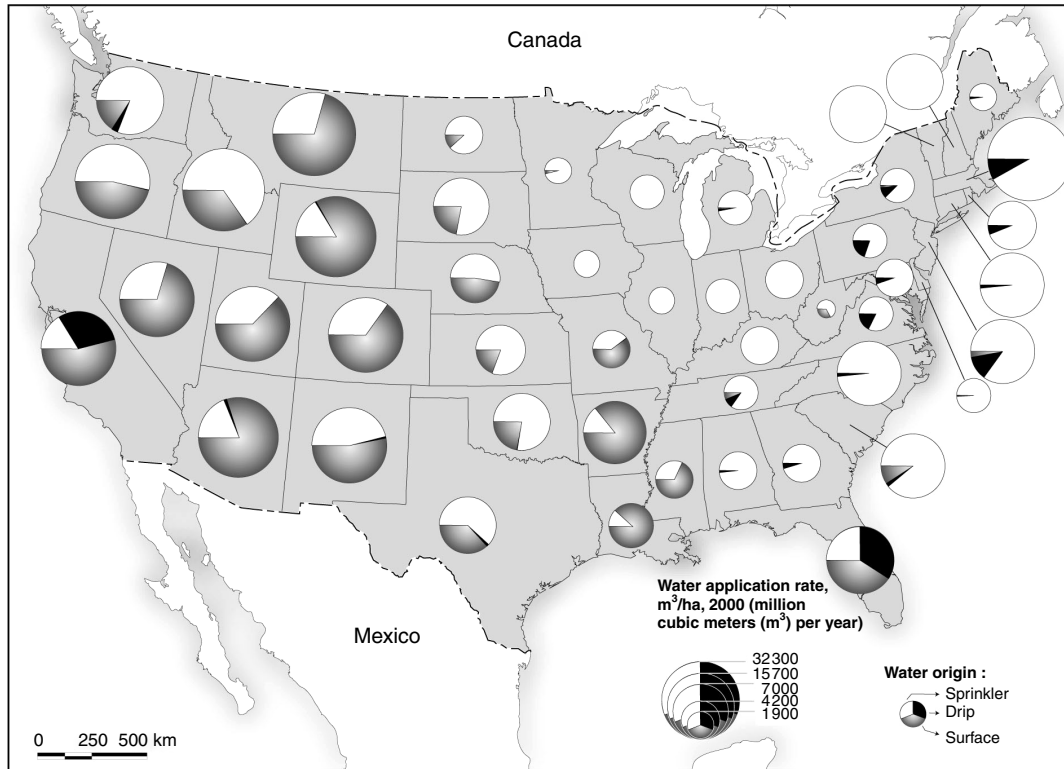


Figure 9. Water application rate, m³/ha, 2000.

used in Florida (34.1% of volume of water spread), Pennsylvania (20%), Virginia (17.9%), New Jersey (12.3%), New York (10.8%), and Massachusetts (8.3%). But west of the Mississippi or in its lower course, surface irrigation is either dominant or holds a significant share of total water used in fields (see Table 7).

Whereas it is well-documented that its water productivity is low (<40%), in water-scarce areas, the extent of

surface irrigation definitely implies improvement potential but points to a major political problem, the very low cost farmers still pay for water carried by expensive water-development projects paid for by all taxpayers in the United States. Fees collected by irrigation districts often do not even pay for the direct costs of moving the water from the dams to the fields, let alone refund the investment.

The low share of sprinkler and drip irrigation in water-scarce areas creates a large productivity improvement potential. Converting a fraction of surface-irrigated fields into sprinkler-irrigated fields could vastly reduce the water application rate. There is a significant relation in the West between a large share of surface irrigation and high application rates. Turning to a more sprinkler-intensive agriculture could reduce water withdrawals and consumption and help relieve the region from the ongoing, bitter, water-sharing debate. But this cannot be done by mere campaigns in favor of more effective watering techniques; farmers are well-aware of them. The issue again boils down to the pricing of water, which is definitely a question that most politicians would not want to tackle directly.

Broadening the topic, these observations underline the structure of water use that agriculture has developed in the United States as of 2000. In the East, a burgeoning and fast-growing irrigation sector relies on water-saving techniques but can use water generously. This water use does not stem from climate constraints, but from financial reasoning, avoiding risk and increasing surface productivity. In the West, water use in irrigation was

Table 7. States Where the Share of Surface Irrigation is Greater Than 30%, 2000

State	Share, % of Total Water Used for Irrigation
Louisiana	88.30
Arkansas	86.03
Wyoming	83.10
Arizona	79.82
Montana	70.93
Nevada	70.48
Mississippi	68.03
Colorado	65.29
Utah	62.41
Missouri	59.55
California	54.16
New Mexico	53.11
Nebraska	47.44
Oregon	46.08
Florida	40.73
Texas	36.83
Idaho	34.67
West Virginia	33.33

the only way to develop intensive agriculture, with crops imported from Europe or the more humid East. But, because of the history of agricultural development, water prices still prevent the adoption of more efficient irrigation techniques. In the context of a slow erosion of irrigation in the West due to rising competition for water from cities and the industry and also due to unsustainable water withdrawal rates in several river basins and/or aquifers, it is important to take into account the improvement potential that agriculture displays in its water use. Massive water diversions are probably not necessary because agriculture, by far the main water user in the West, can reduce its consumption by turning to more efficient irrigation systems. But this comes at a cost.

Acknowledgments

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IRRIGATION WELLS

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In arid and semiarid areas, rainfall is markedly variable in magnitude, time of occurrence, and spatial distribution. Irrigation is required to meet the demand of agriculture, but surface water supplies are also characterized as scarce and unreliable. Consequently, farmers consider it necessary to create a dependable water supply, which is adequate and in a desired location. This management option has spurred farmers to develop the technology of irrigation wells.

History started with the development of irrigation wells to be used for lifting water from rivers (surface water), surface storage (rainfall), and subsurface dug holes (underground water). The discharge rates of such irrigation wells depend on the capacity of the lifting devices.

As pumps were introduced, a boom in another kind of irrigation well occurred to exploit underground water for irrigated agriculture. Promising technologies of irrigation wells include tube wells, skimming wells, dug wells, and radial wells. The discharge rates of these irrigation wells depend on the capacity of the lifting devices and also on the hydrogeologic characteristics of the aquifer.

A tube well has a single strainer that penetrates vertically into the aquifer and operates under suction to extract groundwater at a discharge rate usually greater than 28 L/s. In unconfined aquifers that have fresh groundwater lenses underlain by saline groundwater layers, these wells are prone to extract groundwater of deteriorated quality due to upward movement of saline groundwater that enters the well screen. The thickness of these fresh groundwater lenses varies from a few meters to more than 30 meters, depending on the recharge source. In the surroundings of canals and rivers, this thickness can be more than 30 meters. The diameter of tube-well strainers is usually greater than 10 cm. The greater the thickness of the fresh groundwater lens, the higher the possibility of extracting greater discharge, which requires a larger diameter pipe for the strainer. This type of irrigation well, however, can extract groundwater without the problem of deteriorated quality provided that the well penetrates at less than 60% of the depth of the fresh groundwater lens and its operating time does not exceed 6 hours in any day (1).

For unconfined aquifers where the thickness of the fresh groundwater lens is less than 25 meters, skimming wells are introduced. This type of irrigation well is multistrainer, spread at some horizontal distance from the suction point, and these strainers are joined to a common pump. Like tube wells, the strainers in skimming wells penetrate vertically into the aquifer. However, designing a skimming well to include four to six strainers of 5 cm diameter that spread at a 1.5-m horizontal distance from the suction point makes the design cost-effective and minimizes operation and maintenance requirements. These strainers should be installed at less than 60% of the depth of the fresh groundwater lens of the aquifer. To reduce the risk of quality decline, its operating time should also not exceed 6 hours in any day. This type of irrigation well also works under suction and can usually extract groundwater at discharge rates less than 28 L/s (2).

Dug wells can be used to extract fresh groundwater from aquifers when thickness of fresh groundwater lenses is less than fifteen meters. Depending on the well diameter, there are two kinds of dug wells found in various parts of the world; small dug wells and large dug wells. In both dug wells, groundwater gathers under gravity, and pumps are used to extract this groundwater. Dug wells are mostly suitable for smallholder irrigated agriculture.

A large dug well has a diameter large enough for workers to enter it for construction and maintenance. This type of irrigation well can be dug as deep as 90% of the depth of the fresh groundwater lens of the aquifer and can still avoid the risk of entering the underlying saline groundwater at the bottom of the dug well (3).

A small dug well, on the other hand, has a diameter too small to enter and is constructed from the ground surface using special tools. The treadle pump, which is a human-powered pump, also uses this type of irrigation well as a source of groundwater. For small-scale irrigation, these pumps were first introduced into Bangladesh in the early 1980s, and they are now making their mark in Africa (4).

Radial wells are also multistrainer wells. Unlike tube wells and skimming wells that have vertically installed strainers, radial wells have horizontal strainers. These strainers facilitate skimming a very thin (even less than 3 meters) fresh groundwater lens as they provide large discharges per unit of drawdown—a desirable feature to avoid upward movement of saline groundwater (5). These strainers (usually of 5 cm diameter) can be joined directly to the sump—a collection dug hole, or they are first connected to collector pipes (usually of 10 cm diameter), and then these pipes are joined to the sump. Groundwater comes under gravity to these horizontal strainers, and pumps are used to extract it from the sump. This type of irrigation well, which is an innovative and modern type of dug well, can also be used to support smallholder irrigated agriculture.

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AGRICULTURE AND LAND USE PLANNING

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INTRODUCTION

Many owners of agricultural lands, particularly those at the urban fringe, face strong pressures to convert

their lands to residential or commercial uses. If these forces of change occurred in an environment of perfectly functioning markets, then there would be no economic problem or failure and no economic justification for public policy in the form of planning. We would simply observe the optimal rate of land conversion to reflect societal preferences and changes in income and population over time. However, the forces contributing to agricultural land conversion are rife with informational and policy-driven distortions in both agricultural and nonagricultural land markets, among them, zoning and other local policies that encourage development and inflate land values beyond the reach of agriculture, high-risk and low-economic returns to commodity production, higher incomes and the provision of health insurance in off-farm occupations, unintended consequences of state and federal farm and nonfarm policies, and improvements in telecommunications and in other infrastructure that mute the cost advantages of urban living. Moreover, agricultural lands provide many types of value to landowners, private citizens, communities, and society at large that are not reflected in the market value of agricultural products or in the capitalized value of those products in establishing land lease and sale prices. As a result of both supply- and demand-side dynamics, the market for agricultural land usually fails to reflect social values. Therefore, public policy in the form of some sort of land-use planning is justified.

The American public values agricultural land for many reasons (1,2), beyond its ability to produce food, fiber, and timber. These include curbing negative impacts of ill-managed growth; providing flood control; improving water quality; maintaining groundwater rechargeability; mitigating air pollution, ozone, and greenhouse gas emissions; providing community buffers or greenways; reducing traffic congestion; and easing the fiscal burden of public services. Such values can be reflected within a community by the relative intensity of alternative development; the opportunity for public goods and services, such as parks and trails; wildlife habitats, scenic views, and recreational opportunities; and proximity to urban dwellers (see, for a variety of sources, Ref. 3).

Public values for agricultural land preservation have translated into public support for establishing agricultural land preservation programs at the local, state, and federal levels. A mixture of both regulatory (4,5) and incentive-based approaches (4,6) have resulted in the permanent protection of nearly 1.5 million acres of agricultural land from conversion to nonagricultural uses (7) and the concentration of millions of more acres in agricultural production through zoning. Agricultural land protection through the various tools of land-use planning is a complex issue and process. This article provides only a glimpse of fundamental approaches and resources available for agricultural land-use planning and preservation.

AGRICULTURAL LAND USE PLANNING ISSUES

What land should be preserved and for what purposes? What level of private and public resources should be made available for fee simple land or development rights

purchases? How should priorities be set to protect critical and sensitive areas with limited private and public funds to achieve the best results? Both private and public institutions must address these questions and concerns.

Agricultural land-use planning and preservation constitutes sensible resource management for sustainable agriculture and more broadly for sustainable development. It also stands on its own as a local, state, or national policy goal to mitigate land fragmentation and haphazard development. Various tools, resources, and model programs are available to facilitate agricultural land-use planning and preservation. Each person and community has unique goals, challenges, and collective knowledge that will determine the appropriate mixture of tools to guide its private and public actions in crafting effective local land-use management and policy (8).

Flexibility and knowledge in local conditions are the most pervasive features of successful programs to preserve lands for multiple functions meeting agricultural operations, managing land use, and achieving environmental, social, and economic vitality objectives (4). Scientists and educators at the land-grant universities provide research-based information, analyses, and educational programs that will help facilitate this process at the local and state levels.

Public programs for agricultural land-use planning and preservation may benefit both landowners and the general public. Agricultural land-use planning and preservation programs may be motivated to preserve production agriculture for national food security or for local food choice reasons. In this case, the following are many of the key characteristics typically considered to prioritize land parcels chosen for preservation:

1. Soil quality and productivity
2. Agricultural infrastructure
3. Farming methods, including conservation practices
4. Unique or critical land quality
5. Imminent threat of conversion
6. Critical size of a parcel for a viable agricultural operation
7. Proximity to other protected land for a critical mass to achieve effectiveness
8. Importance to local agricultural and economic vitality (9,2).

Alternatively, the public may be more concerned with issues related to urban growth pressure or preservation of ecological integrity and cultural resources (10). They may perceive preserving agricultural land as a way to sustain community values, lifestyle, or its natural and cultural heritage. In this case, land preservation programs may rank land parcels by the ability to provide direct environmental or economic benefits of open space and scenic beauty, economic opportunity, such as agro-tourism, wildlife habitats, community buffers or greenways, source water or watershed protection, clean air or carbon sequestration, historical or cultural significance, recreational opportunities, such as hunting, curbing current and anticipated development pressure, and/or

complementary community practices, such as agricultural zoning and growth management (1,11).

NATIONAL, STATE, AND LOCAL AGRICULTURAL LAND PLANNING AND PRESERVATION PROGRAMS

U.S. Department of Agriculture (USDA)

Public interest in agricultural land protection, especially against fragmentation and haphazard development, has increased steadily since the 1950s. As this interest has grown, the preservation toolbox has expanded from local and state to the federal level. USDA programs designed to preserve working agricultural landscapes include the Farm and Ranch Lands Protection Program (FRPP), the Forest Legacy Program (FLP), and the Grassland Reserve Program.

The FRPP (www.nrcs.usda.gov/programs/FRPP) leverages federal funds with state and local funds to purchase conservation easements on prime and locally important or unique land by limiting conversion to nonagricultural uses. The FLP (www.fs.fed.us/spf/coop/programs/loa/flp.shtml) establishes partnerships between the Forest Service (USFS) and state forestry agencies to protect environmentally important private forestlands from conversion to nonforest uses. Through conservation easements or rental agreements, the GRP (www.nrcs.usda.gov/programs/GRP) protects and restores grasslands from conversion to cropland and other nonagricultural uses and enables viable ranching operations to restore plant and animal biodiversity.

Many state and local ranking systems are based on variations of the federal Land Evaluation and Site Assessment (LESA) system (www.nrcs.usda.gov/PROGRAMS/lesa/index.html). LESA encompasses a portfolio of land attributes to estimate the relative value of agricultural land. LESA provides a framework to measure the merits of keeping individual parcels in agricultural operation based on local preservation objectives. LESA recently has been tied to GIS technology to facilitate mapping and planning scenarios.

State and Local Programs

Over the past 50 years, public organizations at the state and local levels have used a variety of tools to preserve agricultural land. Examples of these tools include differential tax assessment programs, agricultural district programs, right-to-farm or nuisance protection laws, transfer of development rights (TDR) programs, purchase of agricultural conservation easements—also known as development rights (PACE or PDR) programs—growth management or urban boundaries and agricultural zoning, among a variety of other tools. Private organizations such as land trusts also have gotten involved, typically by accepting donated conservation easements or facilitating the process for landowners to sell easements to a public program.

Beginning in 1956 with the inception of Maryland's differential property tax assessment program, today every state except Michigan has a differential property tax assessment program that assesses agricultural land based

on its use value, instead of its fair market value. The program either reduces property taxes on agricultural lands or defers taxes for as long as the land remains in agriculture. Michigan, Wisconsin, and New York also have programs to allow farmers to claim income tax credits to offset their local property tax bills.

California enacted the first agricultural districts law in 1965, commonly known as the Williamson Act. Landowners are allowed to create "agricultural preserves" areas. New York was the first state to create a comprehensive agricultural district program to protect farmland and support the farming business. Agricultural district programs are authorized in 16 states (7) to create legally recognized geographic entities where agricultural activities and their land bases are encouraged and protected. In addition, all 50 states have nuisance protection statutes (also known as "right-to-farm" laws) for agricultural operations.

In 1967, Boulder County in Colorado (www.co.boulder.co.us/lu) established a TDR program to protect agricultural lands and open space. This program enables landowners to transfer the development rights on one parcel of land to another parcel of land, such as from an agricultural zone to designated higher density development areas. Besides maintaining working agricultural landscapes, TDR programs may be designed for multiple purposes, such as to conserve environmentally sensitive areas or preserve historic landmarks. As of 2000, Montgomery County in Maryland had more than 40,000 acres, which accounted for 60% of the national total, enrolled in the TDR program (7).

In 1974, Suffolk County in New York enacted the first PACE (also known as purchase of development rights, PDR) program. King County in Washington and the states of Maryland, Massachusetts, and Connecticut quickly followed suit. PACE programs are voluntary on the landowner's part and permanently protect agricultural land by removing the development rights. As of 2004, PACE programs operate in 23 states, including 19 statewide and more than 45 local programs (7). Five additional statewide programs have been authorized to acquire easements but have not purchased any to date. In particular, local PACE programs are expanding probably because of incentives from the federal FRPP, which provides matching funds to support local, state, and private easement acquisitions, as well as from landowner and community interest in saving land for agriculture. From 2000 to 2004, the number of local PACE programs increased 32% (7).

Programs at the local and state levels take a variety of forms. Communities need to use a combination of tools and techniques as part of a comprehensive effort to plan for appropriate land use and protect agricultural land. Examples of other common tools that communities use include comprehensive planning; agricultural zoning; limitations on infrastructure, regional tax sharing; urban growth boundaries and growth management; and agricultural viability and economic development programs. Information on conservation easements, private conservation and donation, and local land trusts can be found at the Land Trust Alliance (LTA) (www.lta.org).

Information on farmland protection and stewardship can be found on the Farmland Information Center (FIC) (www.farmlandinfo.org), which has a series of fact sheets on farmland protection tools and an extensive collection of local and state land-use and land preservation laws, statistics, and literature.

Examples of statewide PACE programs include California (www.consrv.ca.gov/DLRP/index.htm), Maryland (www.malpf.info), Michigan (www.michigan.gov/mda/0,1607,7-125-1567_1599--,00.html), Pennsylvania (www.agriculture.state.pa.us/farmland/cwp/view.asp?a=3&q=122454&agricultureNav=|4451|), and Vermont (www.vhcb.org/conservation.html). Prominent local PACE programs can be found in Boulder County, Colorado (www.co.boulder.co.us/openspace); Montgomery County, Maryland (www.mc-mncppc.org/legacy_open_space/index.shtm); and Lancaster County, Pennsylvania (<http://www.co.lancaster.pa.us/planning/site/default.asp>).

Land-Grant Colleges and Universities (LGUs)

Many programs related to agricultural land preservation may be found in the Land-Grant University System, including general national level programs in applied research and outreach education, specific endowed chairs in land-use and planning, and regional or state level programs addressing land-use policy and planning.

Each U.S. state and territory has at least one land-grant university (LGU) (www.csrees.usda.gov/qlinks/partners/state_partners.html), including state agricultural experiment station(s) and a state cooperative extension service. Both the agricultural experiment stations and the nationwide Cooperative Extension Service (www.csrees.usda.gov/Extension/index.html) have a long history of engagement in research, teaching, education, extension, and outreach on issues surrounding land use and policy. Information on land-use management and policy (www.csrees.usda.gov/nea/nre/in_focus/ere_if_nererd.html) can be found on the CSREES website. Researchers at the USDA's Economic Research Service (ERS) (www.ers.usda.gov/Topics/view.asp?T=103610) also conduct economic research on land-use policy issues.

The increasing importance and complexity of land-use issues have led to the establishment of two endowed chairs at LGUs: (1) the C. William Swank Rural-Urban Policy Chair at the Ohio State University (aede.osu.edu/programs/Swank); and (2) the John Hannah Distinguished Professor in Land Policy at Michigan State University (www.landpolicy.msu.edu). Both land-use endowed chairs provide substantial leadership at the state and national levels to address these complex issues through research, education, extension, and outreach (12).

Examples of land-use programs at LGUs include the University of California at Davis (aic.ucdavis.edu/research1/land.html), Colorado State University (dare.agsci.colostate.edu/csusagecon/extension/pubstools.htm#LandUse) (13), Michigan State University (ntweb11a.ais.msu.edu/luaoe/index.asp), the Ohio State University (landuse.osu.edu/), Pennsylvania State University (www.cas.nercrd.psu.edu/Land_use.htm), Purdue University

(www.ces.purdue.edu/anr/landuse/), University of Wisconsin (www.uwsp.edu/cnr/landcenter/), and University of Wyoming (www.uwyo.edu/openspace/).

Nongovernmental Organizations

For landowners, one of the most common ways to permanently preserve their agricultural land is to donate a conservation easement to a qualified conservation organization or public entity. If donating the entire easement value is not feasible, landowners can make a partial donation. According to the LTA (www.lta.org/aboutlt/census.shtml), more than 1260 local and regional land trusts were operating in the United States in 2000 and had collectively preserved more than 6.2 million acres of land and other natural and cultural resources. Information about local and state growth management, land-use planning, and agricultural land preservation in general can be found at the American Farmland Trust (AFT) (www.farmland.org/), the American Planning Association (www.planning.org/), the Smart Growth Network (www.smartgrowth.org/), the Sprawl Resource Guide (www.plannersweb.com/sprawl.html), and the Lincoln Institute of Land Policy (www.lincolninstitute.edu/index-high.asp).

Increasingly, in partnership with local, state, or federal programs, many nongovernmental organizations (NGOs), especially land trusts, work to preserve agricultural lands and open space. Until recently, except in California, most NGOs accepted donated conservation easements. As the 2002 Farm Bill made qualified NGOs eligible to receive federal funds, land trusts have since increased in purchasing agricultural conservation easements. Most land trust activities take place at the local and state levels, such as the Colorado Coalition of Land Trusts (www.cclt.org/) and Sonoran Institute (www.sonoran.org/index.html). However, prominent national organizations working on land preservation include the AFT (www.farmland.org/), the LTA (www.lta.org/), and the Trust for Public Land (www.tpl.org).

CONCLUDING REMARKS

Agricultural land policy and protection is a pressing issue of public concern. The failure of land markets to reflect the social value of extensive land uses relative to intensive uses and the irreversibility of land-use intensification decisions create a particularly fluid and pressing policy environment for lay people, land-use practitioners, and researchers alike. Federal, state, and local governmental agencies and nongovernmental organizations of various scales are engaged in land-use planning as are researchers and outreach educators at many Land Grant Universities throughout the United States. In this article, we have provided a brief overview of the issues and resources and ongoing programs directly influencing agricultural land protection and land-use policy. We have taken pains to point to many of the resources currently available on the Internet, recognizing that this is an area of inquiry characterized by rapid change and development. As such, traditional print

resources, although valuable, cannot be expected to keep pace with the inchoate informational environment found within the agricultural land-use planning and protection arena. Interested readers are encouraged to consult the prominent individual practitioners, program managers, and researchers in the field to benefit from the most up-to-date information available.

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WATERLOGGING

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According to Clayton, land is waterlogged if the water table is within ± 150 cm of the natural surface. A modification to account for the yield of the crop was made and decided to term a land waterlogged if the yield is reduced to 25% of the optimum value due to a rise in the subsoil water. A committee constituted by the Central Water and Power Commission (India) defined waterlogging on the basis of a definition proposed by Kuntze. According to this definition, an area is said to be waterlogged when water stagnates on the land surface or the water table rises to an extent such that soil pores in the root zone of crop become saturated, resulting in restrictions in the normal circulation of air, a decline in the level of oxygen, and an increase in the level of carbon dioxide. Due to its simplicity, the definition proposed by Clayton has been adopted in many countries, and apparently it accounts for both surface and subsurface waterlogging. Thus, the word waterlogging is used to designate stress due to water stagnation or a shallow water table because similar conditions could occur as a result of any of the two causative factors. However, to designate the problem of water stagnation on the land surface, besides waterlogging, surface stagnation and flooding have also been used synonymously. Deep submergence depths caused by floods are generally excluded from waterlogging because the causative factors as well as the solutions to such a problem could be different.

VISUAL OBSERVATIONS FOR PROBLEM IDENTIFICATION

The assessment of the degree and extent of waterlogging can usually be made through simple observations, interviews with local officers of the agricultural and irrigation departments, discussions with agricultural technicians and farmers, through communication, and possibly some field measurements. Field reconnaissance is necessary to find out if the problem exists. Look for the following to arrive at a reasonable conclusion.

- Standing water or wet spots in parts of the field where the crop stand is poor. Standing water in low spots after a prolonged dry period is a useful indicator of the problem.
- Observe wells, gravel pits, and deep channels, which show the depth to groundwater. In India, the water table in open wells is commonly measured and reported. If there are few wells, then install pits or auger small observation wells into the soil to depths of 1–2 m below the soil surface. If soil horizons are reached which are gley, wet, and may contain black or red mottles, one can assume that soils are poorly drained at this level (1).
- Vegetative cover is a good index to the depth to the water table in many areas. Certain grasses/crops and trees are more common in waterlogged lands. It is

a useful practice to look for these features while surveying. Trees such as willows, cottonwood, and poplar often thrive in high water table areas. Reed grass and sedges are also common.

- A general yellowing of the crop is noticed in areas that are affected by surface stagnation or have a high water table.
- Salts in the form of a white crust on the soil surface reveal the problem of waterlogging. Since the problem could occur even when the crust is not present, its absence does not mean that there is no problem of waterlogging.
- Absence of surface drains or their condition (full of vegetation or plugged up) could also indicate a problem of waterlogging/drainage.

FIELD MEASUREMENTS TO ASSESS THE PROBLEM

In the field, waterlogging is measured/assessed through physical measurement of the depth to the water table using open wells, tube wells, or observation wells. The depth to the water table in the range of 0–1.5 m would usually indicate a problem of waterlogging, although it would depend upon the soil and crop characteristics. A relatively shallow water table may not be a problem in a coarse-textured soil, but even a relatively deeper water table could cause a problem in a fine-textured soil.

In the field, the oxygen diffusion rate (ODR) is a good measure of oxygen deficiency. ODR is measured with an oxygen diffusion rate meter. Measurements are usually made at a 10-cm depth replicated at 5–10 locations. Stolzy and Letey (2), in a review on plant response to measured ODR, concluded that the roots of many plants do not grow in soils with ODR values of $20 \times 10^{-8} \text{ g cm}^{-2} \text{ min}^{-1}$. For germination and emergence of seeds, minimum ODR values of the order of $40\text{--}80 \times 10^{-8} \text{ g cm}^{-2} \text{ min}^{-1}$ are mentioned. The values of ODR that indicate adequate aeration are in the range of $20\text{--}35 \times 10^{-8} \text{ g cm}^{-2} \text{ min}^{-1}$ (2).

The oxidation–reduction potential (redox potential, RP) of the soil is also used as an indicator of the problem of waterlogging. In practice, the RP of the soil is measured with an oxygen meter using an Ag–AgCl reference electrode. The readings can be converted relative to the standard hydrogen electrode by adding 222 mV. As long as O_2 is present in a significant amount, the RP remains high (800–900 mV). The oxidation–reduction potential for reduction of NO_3^- to NO_2^- is 420 mV at pH 7 and 530 mV at pH 5. Nitrate is readily reduced to N_2O or N_2 at these oxidation–reduction potentials.

CAUSES OF WATERLOGGING

A large number of factors combine to create the problem or affect the degree of the problem of waterlogging. Geomorphic features like palaeochannels, local depressions, lower elements of slopes, and flat flood plains are invariably subject to varying degrees of waterlogging. Unfavorable topography of the terrain has a profound effect on the magnitude of the problem. Climatic conditions such as

heavy rainstorms, a characteristic of monsoon climatic regions, coupled with poor natural drainage and/or poor upkeep of the drainage system result in surface stagnation of water. Development activities such as construction of roads, bridges, railway lines, and buildings result in choking of natural drainage thus aiding surface stagnation and a rise in the water table. Soil texture and structure having direct bearing on water storage and transmission could cause surface stagnation of water. Formation of a hard layer at plow depth in rice–wheat rotation prolongs the surface stagnation of water, especially in areas of poor on-farm water management. Prolonged heavy precipitation and a high water table resulting in reduced absorption of rainwater is a sure prescription for surface water stagnation. Increasing encroachment of wetlands for developmental activities compounds the situation.

Many factors that result in surface stagnation aid the buildup of the water table in a region. However, a rise in the water table is an inevitable consequence of introducing surface irrigation through the interbasin transfer of water. Several factors together determine the rate at which the water table would rise in a given setting. Sources of excess soil water that result in high water tables include high precipitation in humid regions, surplus irrigation water and canal seepage in irrigated lands, and artesian pressure. The geomorphologic setting of an area and the presence of impervious substrata/hard pan such as clay pan, calcic layer, duripan, or fragipan could obstruct the downward movement of water and cause short-term waterlogging. If such conditions persist for long, the land might develop a perched water table.

EFFECTS OF WATERLOGGING

Both air and water in appropriate proportions are essential for plant growth. Disturbance of the delicate balance between these two components as a result of waterlogging results in poor aeration. Except for winter conditions, an air-filled porosity of less than 10% is considered to indicate inadequate aeration. Because gases in general, diffuse about 1000 times more slowly in water than in air (3), this causes an imbalance in the proportion of oxygen and carbon dioxide in the root zone. This imbalance manifests its effect on the physiological processes of the plants and on the nutrient status of the soils. The chemistry and microbiology of submerged lands is quite different from the chemistry and microbiology of nonsubmerged lands (4). This can affect the availability of many nutrients. For example, nitrogen can undergo denitrification more readily and be lost to the atmosphere as a gas. Leaching of plant nutrients could also result in their nonavailability to plants. The anaerobic (reducing) environment results in changes that can result in deficiencies (reduced uptake) or toxicities (excess). For example, the increased availability of silicon apparently decreases the uptake of iron, whereas sulfide, ferrous, and manganese ions accumulate in waterlogged soils. The important inorganic and organic compounds that are produced in submerged soils are iron, sulfides, and organic acids. Ponnampetuma (4) reported that the level of ferrous iron in submerged soils often exceeds 3000 ppm. Such levels could be toxic to rice plants,

although the degree of toxicity may depend upon the pH and the presence of hydrogen and ferrous sulfide. Soil sulfate under highly reduced conditions as might occur in submerged land is reduced to H_2S . Some of the organic acids such as acetic, propionic, and butyric are found in submerged soils; acetic acid is the most abundantly available carboxylic acid. These acids are toxic to rice seedlings.

All biological processes are strongly influenced by soil temperature. Waterlogged soils have a large heat capacity and therefore, are relatively colder than dry soils. Crop growth in waterlogged soils, therefore starts later and is slower than in dry soils.

The unconfined compressive strength of a soil is exponentially related to the moisture content. Water content is the chief factor determining the strength and also the deformation characteristics of cohesive soils. As a result, dry soils can support heavy loads because of the rigidity of the individual soil particles and the high internal friction of the soil mass. At high moisture content, the soil is susceptible to plastic deformation and consolidation. With increasing soil moisture, it loses its load-bearing capacity altogether. As a result, heavy machinery cannot be employed on waterlogged lands. A direct consequence of this would be delays in the sowing/harvesting operations of crops. To manipulate the soil without compaction or puddling, the moisture content of the soil must be below some critical level, depending upon the soil type. Compared with sand, clays have a narrow range of water content in which tillage will readily break the soil. The timeliness of the operation significantly affects the yield of crops in humid regions and in soils that have high clay content. Even the quality of the crop is affected because of the delay in harvesting.

Besides what has been stated before, excess water and the resulting continuously wet root zone can lead to some serious and fatal diseases of the root and stem (5).

It is now apparent that the adverse effect of temporary waterlogging depends not only on plant species but also on the physiological stage of growth, the time and duration of waterlogging, the light intensity, the temperature, and the fertility of the soil. Injury to crops is particularly severe when crops are flooded on hot days. The ill effects are generally greater if the waterlogging conditions are accompanied by salinity/sodicity. It is now known that no single factor affects crop yield in waterlogged soils. Generally, it is a combination of poor aeration, reduced uptake of plant nutrients, imbalance between the uptake of various ions, and the toxic effects of various elements/cations.

The ill-effects of surface stagnation of water and a high water table extend beyond the agricultural sector. The anticipated adverse effects of waterlogging are listed in Table 1.

SURFACE STAGNATION AND CROP YIELDS

Excess water in the crop root zone is injurious to plant growth. Crop yields are drastically reduced on poorly drained soils, and in cases of prolonged waterlogging,

Table 1. Anticipated Losses due to Waterlogging

Agriculture	Landscape and Infrastructure	Socioeconomic
Decline in agricultural production	Damage to infrastructure	Increased socioeconomic disparity
Restriction on crops	Landscape degradation	Increased expenditure on health related services
Decline in product quality	Decline in ecosystem health	Migration from rural to urban areas
	Damage to soil health	Increased gender disparity

plants eventually die. Crops vary in their tolerance to surface stagnation. Considerable variation in waterlogging tolerance exists between and within species (6,7). Some crops thrive if water stagnates on cropped land. Lowland rice is one such fine example. However, most dry foot crops are adversely affected, and their yields are reduced. The late Arthur Hodgson, considered a world leader on the topic, extensively and intensively studied the effect of surface stagnation of water on cotton yield. Williamson and Kriz (8) reviewed experiments on flooding and waterlogging. Their conclusion is that when plants are actively growing, the severity of injury depends on the growth stage and the time of the year at which flooding takes place. The influence of the latter is closely related to soil and air temperatures. The Indian experience, however has been that in semireclaimed alkaline lands, surface stagnation at an early growth stage is relatively more harmful than at later growth stages. It might be due to the fact that most crops are sensitive to salts at early growth stages.

Considerable work has been reported on the flooding tolerance of rice (9–11). The extent of damage or yield reduction as a result of water stagnation depends on the crop tolerance, crop growth stage at which water stagnation occurs (12–15), duration of water stagnation/flooding (16–18), soil type, air and soil temperature, and other agroclimatic conditions. Studies have also been conducted on the imbalance between the uptake of various ions to identify the causes of yield reduction (19–22) and amelioration of adverse effects through additional doses of nutrients (21,23,24).

Because crop conditions vary over time, it would be essential and useful to integrate the degree of surface stagnation and duration to assess adverse effects. Some limited attempts in this direction have been made. In spite of their limitations, these concepts could serve as starting point for more elaborate investigations.

To integrate the effect of excess water and time on crop yield, tomato yields were related to the number of days that the soil is >10 mm wetter than field capacity. It has been shown that, on average, the yield declined by 0.26% for every day that the soil is >10 mm wetter than field capacity (5,25).

Gupta and Pandey (26), following Sieben (27), integrated the effect of depth of surface stagnation on the yield of rice. According to their hypothesis, rice yield might not suffer due to occasional high depths of water stagnation. However, at some time, the yield might start decreasing. According to this hypothesis, the stress due to excess water depth, Ewd, could be expressed as follows:

$$\text{Ewd} = \sum_{i=1}^n \sum_{j=1}^m (D_{ij} - d) P_{ij} F_j \quad \text{for } D_{ij} \geq d \quad (1)$$

in which Ewd is the depth day index, cm days; D_{ij} is the depth of stored water at the i th depth interval and j th crop stage, cm; d is the optimum depth of ponding, cm; P_{ij} are the number of days for which submergence occurs in the j th growth stage, days; m is the number of growth stages; n is the number of days in a given growth stage; and F_j is a constant depending on the stage of growth of a given variety. Using the data reported by Gupta and Pandey (28), it was observed that the crop yield decreases only to the extent of 6.4% at a depth day index of 250 (26). The yield is not reduced until the depth day index is 150. This technique is a powerful tool to work out the severity of waterlogging as well as the actual time to carry out drainage.

Gupta et al. (29), while reviewing the Indian experience in crop yield influenced by the duration of water stagnation, attempted to develop a crop production function as affected by duration of water stagnation. It was found that a piecewise linear model of the same nature reported by Maas and Hoffman (30) fit the studied data sets well. Crops have a threshold duration up to which there is no yield decline till the threshold is reached; beyond that, there is a linear decrease in the crop yield till the duration at which the crop yield is likely to be zero. Gupta et al. (31) confirmed the validity of the model while comparing the production functions for three winter crops. The duration of water stagnation at which decline in yield would be 50%, is 7.2 days for wheat, and it is 7.9 and 12.2 for mustard and barley.

GROUNDWATER DEPTH AND TOLERANCE OF CROPS

Per se, a shallow depth of groundwater is not harmful to plants. Indirectly, it determines the prevailing moisture conditions and therefore has an influence on water supply, aeration, and the heat properties of soils. As such, the yields of most crops are affected when groundwater is shallow. As with surface stagnation, crops also vary in their tolerance to a high water table (Table 2). Lowland rice can thrive on soils that have a shallow water table.

Studies on the III-Effects of a High Water Table

Numerous laboratory and lysimeter/field experiments on the effect of water table depth on crop yields have been conducted at various locations. Lewis and Wor (32) reported that orchards in Oregon require, ideally, a water table depth of 180–250 cm, but a higher water table persisting for 3 to 4 days following rain or irrigation does no harm to the plants. Deep-rooting fruit trees tend to suffer most when the water table is close to the surface such that it helps establish a shallow rooting system. Roe

Table 2. Tolerance Levels of Crops to High Groundwater Table (Groundwater at 50 cm)

Tolerance Level	Crops
High tolerance	Sugarcane, potatoes, rice, willow, plum, broad beans, strawberries, some grasses
Medium tolerance	Sugar beet, wheat, oats, citrus, bananas, apple, barley, peas, cotton, pears, blackberries, onion
Sensitive	Maize, tobacco, peaches, cherries, olives, peas, beans, date palm

Source: Internet Search.

(33) reviewed the work on yields of vegetables related to water table depth in the eastern United States. The general finding is that a water table from 30–60 cm from the surface is required for most vegetable crops. Penman (34) observed that citrus trees in Australia, remained in healthy condition for the first 8–10 years where a water table was within 120 cm from the surface. Adverse effects appeared thereafter.

Williamson and Kriz (8) provide a good account of the tolerance of crops to a constant water table. The data given agree fairly well with values mentioned by Williamson and van Schilfgaarde (35), Williamson and Carrekar (36), and Feddes (37). In these papers, required water table depths of 60 to 90 cm for sandy soils and 100 to 150 cm for clay soils are mentioned; the depth within these ranges depends on soil type, crop, and climatologic conditions. The relative tolerance of crops to waterlogging has also been listed according to the cause of damage (8). It is now known that the responses of agricultural crops vary genetically and physiologically to the water regime and therefore, crops that are not adapted to a high water table receive a severe setback when roots encounter a high water table. The extent of damage depends upon the crop, the crop growth stage at which high water table conditions occur, the duration of the high water table, the soil type and agroclimatic conditions.

Visser (38) described an intensive survey of prevailing groundwater table depths in The Netherlands. A general crop production function was described as (1) an increasing yield as depth increases at shallow depths, (2) a certain optimum range, and (3) a yield decline at greater groundwater depths. At shallow depths, the crop suffers from lack of aeration, but at greater depths, water deficiency is the cause of yield depression. This kind of crop production function is expected when more or less the same amounts of irrigation are given to crops.

Though accepting this logic, it is now known that the contribution of the water table to meet a part of the crop water requirement should not be overlooked (39,40). By properly scheduling irrigation, the shape of the crop production function could be altered significantly. The actual experience and field results substantiate the observation that using careful management, crop production can be increased even when the water table is shallow. For an equivalent yield of a wheat crop under a different water table, the amount of irrigation needs to be decreased as water table depth decreases. For example,

Pandey and Sinha (41) observed that an equivalent yield of wheat could be obtained when the number of irrigation were varied from 0 to 4 as the water table was lowered from <75 cm to 150 cm. No hard and fast rule can decide the optimum depth to the water table for a given crop. The information given in the literature should be treated only as a guideline. To take advantage of the shallow water table contribution to the crop water requirement, controlled drainage has been perfected. It has been shown that controlled drainage helps to reduce water stress at least in low rainfall years and more N use by plants (42,43).

Under field conditions, soil–plant systems vary continuously; evaluating the effect of water table on plant growth would require integration of water table depth over time during the entire growing season.

Sieben (27), when reviewing the results of experimental drainage fields in the Dutch Ysselmeer polders, drew special attention to the influence of high fluctuating winter water tables. He took 30 cm below surface as a critical level and computed so-called SEW₃₀ values from

$$SEW_{30} = \sum_{i=1}^n (30 - x_i) \quad (2)$$

where x_i are daily water table depths below the surface. For this computation only x_i values smaller than 30 are taken into account, so that the sum is a measure of the excess over the 30-cm level. Large SEW values generally indicate poor drainage conditions. It seems that at SEW values of 100 to 200, there is a decrease in yield. The analysis also revealed that the tolerance of four crops is in the order of winter wheat > peas > oats > winter barley. The yield of wheat is not adversely affected till this index passes the 200 mark, whereas barley is tolerant to waterlogging only up to an index of 120.

Questions of the validity of this concept have been raised mainly on the basis that identical SEW values do not necessarily imply identical conditions. It is also argued that the concept does not account for the stage of growth at which shallow depths to the water table occur (44). Although, it should be possible to modify the Sieben Eq. 2 to avoid such criticism, yet the nonavailability of data on crop growth stage dependent factors would usually make such modification an academic exercise. A recent attempt to determine crop susceptibility factors for five growth stages for corn and soybean could provide answers to such a criticism (45). These authors also concluded that normalized crop susceptibility factors for wet stress might be roughly approximated for some crops from existing data on yield response to dry stress.

Comparative Assessment of Stress Due to Surface Stagnation and a High Water Table

Several experimental results exist in the literature where adverse effects on crop performance at the same site due to surface stagnation as well as a high water table have been studied (19,46). However, in many studies, it has not been possible to compare the results, as the stress levels cannot be compared. Ahmad et al. (46) however, imposed equivalent stress levels based on Equation 2, as

proposed by Sieben (24). At the same stress level due to submergence and depth to the water table of 15 cm, crop vegetative growth and shoot uptake of N, P, and K were significantly less for submergence than due to shallow depth to the water table.

MANAGEMENT AND RECLAMATION OF WATERLOGGED LANDS

Waterlogging is a widespread problem under different agroclimatic conditions, so several options have been field tested to minimize the adverse effects of waterlogging. Among them are land forming (bed plantation, raised and sunken beds); crop selection in favor of tolerant crops; skipping or delaying irrigation so that crops can draw a part of their water requirement from the shallow water table; applying less water per irrigation; cultural practices such as hoeing and weeding, including chemical control of weeds; application of additional doses of nutrients through soil or foliar application, and mulching to minimize secondary salinization. However, note that these are short-term measures and might or might not be useful in each and every case. Therefore, improvement in drainage seems to be essential, as it would be the only permanent solution to the problem.

The lack of drainage or the inadequacy of it is rapidly becoming a major constraint on agricultural production. The productivity of agricultural lands can be sustained only if drainage improvements are undertaken on cropland currently affected by submergence or high water tables. Very often, the natural drainage in an area along with good water management is sufficient to eliminate excess water and to preclude the need for drainage systems. However, there are many situations where surface drainage would be required. In humid regions as well as in monsoon climatic conditions where rain storms of 100 mm or more in one day can be anticipated, even in arid and semiarid regions, surface drainage is an essential prerequisite to avoid water stagnation. Farmers who have heavy textured soils, soils with a plow layer that develops in lowland rice-wheat systems, alkaline lands with poor water absorption characteristics, or those who rely mainly on surface irrigation should have adequate surface drainage facilities to remove excess water. A uniform slope of about 1:1500 is desirable to drain irrigation water or rainfall off a field.

If recharge to the groundwater is more than the discharge and is natural drainage is unable to take care of this recharge, such a situation calls for subsurface drainage. Subsurface drainage may be accomplished either through the construction of open trenches or through buried clay or concrete tiles or perforated pipes. It must, however, be realized that most crops have an optimum depth to the water table for optimum crop yields. If the water table were lowered beyond this depth, there would not be any significant effect on the yield of the crops, and additional investment in intensive drainage would not be beneficial.

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WATER QUALITY MANAGEMENT IN AN AGRICULTURAL LANDSCAPE

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INTRODUCTION

Water quality is important for a wide range of reasons including human health, food production, recreation, and biodiversity. Globally, agriculture exerts big demands and impacts on water supplies. In arid areas where groundwater may be the significant or sole source of water, agriculture can have profound effects on the rate and composition of groundwater recharge (1), while in tropical areas, high annual rainfall levels and the demands of tropical crops mean that water quality concerns are dominated by agrochemicals, untreated wastewater, and eroded sediment (2). In temperate zones, moderate rainfall throughout the year means there are few periods when agricultural activities can take place without some risk to water quality and, consequently, temperate water resources are also at constant risk of contamination from agriculture.

The range of agriculturally derived pollutants that may compromise water quality is listed in Table 1. Delivery of these contaminants to water may be associated with surface water runoff or with the movement of eroded soil particles. Contamination of water in agricultural landscapes also occurs directly, such as when grazing animals enter watercourses for access, drinking water, or shade. The contamination of water has widespread implications for human and animal health, for plant and animal biodiversity, and for safe and economical water industries, including water abstraction, fish farming, and water sports. Because of these implications, the protection and maintenance of water quality is legislated both nationally and internationally. In the following sections, the sources, mechanisms, and mitigation of agriculturally derived threats to water quality are described.

NITROGEN (N)

Nitrogen, as nitrate, is soluble, is retained inefficiently in soils, and is only weakly adsorbed by eroded sediment.

Table 1. Pollutants of Concern Within Agricultural Landscapes (5)

Category	Main Pollutants	Example Sources	Example Environmental Impacts
Soil	Silt, phosphorus, pesticides	Soil cultivation and drainage, overgrazing, outdoor pig systems; dairy and beef herds	Silting of river bed gravels damaging fish spawning, invertebrate and plant habitats; excessive turbidity affecting plant growth and fish behavior; excessive sedimentation affecting rooted plants; economic costs to water abstractors; acts as carrier of other pollutants such as P and pesticides
Slurry, manure, dirty water	Organic matter, nutrients, pathogens, ammonia, metals	Slurry and manure storage and application to land; excretion directly to land; runoff from farm yards and buildings; irrigation	Deoxygenation and eutrophication, affecting all aquatic life; blooms of toxic blue-green algae; carrier of pathogens and pollutants
Fertilisers	Nitrate, phosphate	Inorganic fertilizers; slurry and manure	Eutrophication causing ecological degradation; increased water treatment costs
Veterinary medicines	Antibiotics, hormones, growth regulators, parasite control	Slurry and manure application to land; sheep dip application and disposal	Toxicity to aquatic species; contamination of potable supplies
Disinfectants	Halogens, organic chemicals, phosphorus	Use in dairies, pig and poultry housing	Toxicity to aquatic species; contamination of potable supplies; eutrophication causing ecological degradation

Runoff from agricultural farms is therefore a major source of N entering rivers, lakes, and coastal waters, particularly in areas of intensive livestock farming where stocking rates are high and feed is imported. Nitrogen levels in waters draining arable land are also high because of fertilizer inputs (surplus N not absorbed by the crops is discharged into groundwater, fresh water, and coastal waters), and because ploughing encourages organic N mineralization [the process by which organic N is converted to plant-available inorganic N (ammonium) by microorganisms]. Subsurface movement and loss of N also increases with sward age and drainage.

Nitrogen pollution of waters has significant detrimental effects. Apart from acidification due to nitrification of ammonium and the toxicity of various forms of N, the most significant impact is that of eutrophication, described as the most important environmental problem facing aquatic ecosystems in Europe (3). The process of eutrophication in surface waters can lead to a decrease in species diversity, oxygen depletion, increased sedimentation, and increases in toxic blue-green algae. Nitrogen also represents a hazard to human, particularly baby, health when present in drinking water supplies.

Preventing N pollution would require shifting from intensive arable and horticultural land use to extensive unfertilized grassland, which is socially and economically untenable. Therefore, the most effective ways to minimize excessive N leaching from agriculture include switching to organic farming (which would reduce N inputs), using minimum cultivation (which would reduce runoff generation), and planting a cover crop (such as rye or mustard) after harvest and before the next crop (which would reduce runoff generation and increase N uptake). Additional beneficial management practices include choosing high yielding crops to maximize uptake of N, calculating

fertilizer requirements making allowance for existing soil N and organic amendments, avoiding unnecessary fertilizer applications, applying fertilizers away from water-courses, maintaining disease-free crops (as these cannot use N efficiently), and practising careful irrigation to avoid runoff. Alternatively, a curative approach of blending water with low N supplies or biological or chemical treatment to encourage denitrification may be employed (4), although this approach is more costly and less effective than preventing N pollution in the first instance.

PHOSPHORUS (P)

In the United Kingdom more than 50% of P entering surface waters is now thought to be derived from agriculture (5). There are three sources of P available in the landscape for mobilization and transfer to water bodies:

1. Soils contain different background levels of P and, therefore, different levels of P sensitivity to natural transfers in both soluble and sediment-laden forms through runoff and subsurface water flows. Transfer rates depend on the parent material, rainfall, and vegetation cover.
2. Their high specific surface area means soil particles detached during soil erosion have high adsorption capacity for nutrients such as P, which results in an enrichment of P in eroded sediments. The rates of detachment and transfer are accelerated by human impacts such as arable farming and livestock grazing.
3. Transfers can occur where P is added to the land in manures and fertilizers, although best management

practices are available to farmers to minimize these effects.

The relative proportions of P loss in sediment-bound and dissolved forms are highly site specific. Sediment-associated P constitutes the majority of P transported in surface runoff from agricultural land (5). However, dissolved P has a greater impact on water quality than sediment-bound P because it is more readily biologically available (6).

Phosphorus can negatively affect the quality of soils, groundwater, and surface water through eutrophication. In fresh waters, where P is considered the main limiting plant nutrient, elevated P loads tend to favor a few species and promote excessive algal growth. Submerged plants are lost and there are consequential impacts on invertebrate and fish species. As well as the diminished conservation status of eutrophic waterbodies, elevated P concentrations have been associated with impaired human and animal health.

To prevent these impacts, mitigation approaches address P sources or transfer mechanisms of both sediment-associated and runoff-dissolved P. Source approaches try to limit soil P content or its susceptibility to transfer, and include applying P based purely on the needs of the crop and the existing amount of residual soil P, or using chemical amendments to alter soil chemistry and hence reduce the solubility of soil P. Adding alum, for example, encourages P to precipitate with Fe (iron) and has been successfully applied to reduce runoff P concentrations (7). The alternative approach to controlling P is to limit transfer by reducing the occurrence and magnitude of transfer agents, especially runoff and sediment production. Measures to do this include increasing ground cover, contour cultivation, and minimum tillage. Uncultivated buffer strips at field edges, constructed wetlands, and ponds are also efficient at retaining P in agricultural runoff, but edge of field P transport can still occur in soils experiencing significant amounts of lateral subsurface flow (such as sandy soils). Also, while buffer strips can contain P bound on sediment, over time P storage within the buffer strip may create an active source of P flushed out in runoff during storm events.

ORGANIC MATTER

Organic matter as a pollutant concerns animal feces and urine (1) derived directly from grazing (unhoused) livestock, (2) deposited within housing or yards and stored, often with soiled bedding, as slurry or manure, or (3) combined with dairy parlor washings, silage effluent, and runoff from livestock yards and housing as dirty water.

Runoff of slurry, manure, or dirty water after application to land can occur with excess application, when heavy rain follows spreading, or where the terrain is too steep or insufficiently vegetated to ensure plant uptake of such applications. Contamination is also more likely where applications of organic material are close to drains and watercourses, where livestock have uncontrolled access to watercourses, and from farms with inadequate or poorly managed dirty water management systems.

Field or farmyard runoff containing organic matter may result in chronic organic enrichment of streams, ponds, lakes, and ditches. The organic material is broken down by bacteria causing deoxygenation of the water, because the biochemical oxygen demand of these materials can be one to two orders of magnitude higher than that of untreated domestic sewage. This loss of oxygen leads, in turn, to loss of invertebrate and fish life. In addition, organic matter from diffuse agricultural sources, especially from slurry and manure, may contain fecal bacteria, including *Escherichia*, *Cryptosporidium*, and *Giardia* species, which, in sufficient quantities, may be harmful or even fatal to human health.

Measures to minimize the risk of organic matter contamination of watercourses can be applied at the source, or during application or transfer stages. Manure can be treated to reduce pollution potential before land spreading, using chemical, biological, physical, or thermal processes. These processes may have additional benefits such as the recovery of nutrients including P (8). Alternatively, diet manipulation can be used to reduce the nutrient content of excreta without compromising animal performance (9). Within the farm area, runoff from animal enclosures and yards requires collection or diversion to storage or treatment areas to prevent runoff to watercourses. This is easier if clean water from roofs is diverted before coming into contact with organic material. Stored dirty water may then be applied to land, so that nutrients can be taken up by the crop. In storage systems for organic material, overflow or seepage can be prevented through careful siting of stores to avoid permeable soils and rock, effective sealing of the storage unit, regular checking and emptying to minimize overflow, and the diversion of surface water or rainwater (10).

Reducing organic matter contamination of water can be achieved by maintaining adequate storage facilities and land for application, maximizing knowledge of manure nutrient content and crop nutrient requirements, and employing best management practices, such as avoiding spreading when the soil is saturated or frozen, and minimizing application to steeply sloping land near watercourses. Applying soil erosion and runoff control measures is also effective in reducing organic pollutant transfer. The longer manure is in or on the soil before crops can use its nutrients, the greater the risk and opportunity for loss of nutrients and contaminants to surface waters. Spreading organic matter in spring or summer ensures maximum use of nutrient content and avoids the risk of spreading on frozen ground or before rainfall. Spreading should be at a rate appropriate to the crop and variable within the field according to slope, morphology, and proximity to watercourses. The pollution potential of grazing animals can be minimized by restricting access to streams. Once applied to land, transport of organic pollutants in runoff and sediment may be hindered by contour cultivation and the use of buffer strips.

SEDIMENT

The detachment and transport of soil particles from the land is known as soil erosion. Eroded soil particles enter

rivers during periods of high rainfall and consequently high surface runoff. Sediment loads depend on land use, cultivation techniques, stocking densities, drainage, soils, slopes, geology, and rainfall. Heavy sediment loads can occur as a result of soil compaction (which may be associated with loss of organic matter, heavy machinery, multiple cultivations, or excessive livestock densities), soil capping, cultivation of inappropriate land (e.g., steep land), concentrated flow pathways along farm tracks and drainage systems, and the removal of natural runoff interceptors (such as hedges and uncropped land).

Sediment is considered a pollutant in its own right and as a carrier of adsorbed pollutants. Sediment itself increases water turbidity and smothers the bed of rivers and lakes when deposited in watercourses. Fine sediment particles clog river gravels, reducing water flow and aeration of the river bed and hence reducing the survival of fish eggs, invertebrates, and plants. Also, through their adsorption onto soil particles and aggregates, the transfer of sediment carries P, pesticides, pathogenic fecal bacteria, heavy metals, and other contaminants into surface waters.

Mitigation of the impacts of sedimentation may be focused at the source, although interception of sediment during transfer is also effective. Full consideration of site soil conditions is vital before any access or agricultural work is undertaken, as is avoiding unnecessary or ill-timed cultivations and access when the soil is wet and most susceptible to degradation. Positive measures to prevent soil erosion include contour cultivation and drainage, minimum or no tillage cultivation, establishing terraces or strip-cropping, maintaining vegetation cover through restricted grazing and the planting of cover crops, and maintaining and promoting soil organic matter content through mulching and incorporation of crop residues. Measures to minimize transport and delivery of eroded sediments to watercourses include employment of uncultivated buffer strips, grassed waterways, and sediment traps to filter suspended sediment from runoff. Sedimentation ponds or constructed wetlands can reduce runoff velocity and encourage sedimentation outside the watercourse system (11).

PESTICIDES AND OTHER CHEMICALS

A pesticide is defined as any substance prepared and used for the destruction of any pest. Pesticides therefore include herbicides, insecticides, fungicides, antifoulants, sheep dips, masonry and timber preservatives, and veterinary medicines. Agriculture and horticulture account for over 80% of pesticide sales, and applications in crop production are the main source of pesticides that may leach (move in water through the soil profile) to groundwater or run off to surface water in dissolved or particulate form. Pesticide characteristics, soil type, cultivation, and drainage all affect pesticide transport to water. Heavy rain after spraying increases the chance of pesticide movement in runoff. Loss may also occur through spray drift and volatilization, and research has shown that a high proportion of pesticides in surface waters may come from very small farmyard spills and drips from handling and sprayers: these often fall onto hard

surfaces, from which they are readily washed into drains, ditches, and watercourses. The use and disposal of sheep dip is a particular risk to fresh waters: water is contaminated by dips located near watercourses and by inappropriate disposal of spent dip. Veterinary medicines and disinfectants used on dairy herds also pose a risk to the aquatic environment.

Pesticides are likely to have toxic effects on aquatic organisms either directly through toxicity or indirectly through changes in the environment. The impacts of pesticides range from rapid destruction of the tissues in contact with the pesticide to slow deterioration in a physiological process. Exposure to sublethal concentrations may cause genetic, physiological, or behavioral changes, including reproduction impairment, inhibition of brain activities, and growth reductions (12).

Pesticides applied to cropland are degraded by microbial and chemical reaction in the soil, temporarily immobilized by adsorption onto soil particles and organic matter, taken up by plants or pests, removed when the crop is harvested, or lost to the environment through runoff in surface water, volatilization into the atmosphere, and leaching to groundwater. For weakly to moderately adsorbed pesticides, the major carrier is runoff, and therefore controlling the loss of pesticides and persistent organic pollutants to watercourses involves reducing runoff. The most important factor affecting the amount of pesticide transported is the amount of pesticide present at the soil surface at the time of precipitation or irrigation. Reducing pesticide pollution can be promoted through the use of natural predators, crop rotations, integrating pesticide control with soil conservation, and using mechanical procedures, such as conservation tillage and vegetative strips (buffers), to reduce runoff and soil erosion and hence the loss of organic chemicals. The way in which the pesticide is applied may also affect pesticide loss. For example, tillage may incorporate the pesticide into the soil, so reducing the risk of pesticide loss by runoff.

HEAVY METALS

Heavy metals (high density elements naturally present in very low abundance in nature) of concern in agricultural landscapes include arsenic, cadmium, boron, selenium, nickel, lead, and mercury. Agricultural use of mineral fertilizers, feed concentrates, and medicines all contribute to farm inputs of heavy metals, although atmospheric deposition may also play a role, as in the deposition of lead and cadmium (13). Heavy metal transfer to water occurs through leaching and through adsorption onto eroded soil particles (preferentially onto clay particles) subsequently deposited within waterbodies (14).

When leached to surface waters, heavy metals are toxic to aquatic organisms (of particular concern is the impact of mercury on fish) and can impact negatively on drinking water quality. Several heavy metals, including mercury and lead, are known to bioaccumulate in the aquatic food chain.

Mitigating heavy metal contamination of water requires amending the source, plant uptake, and transfer of heavy metals. The choice of soil amendments (e.g.,

manure, compost, sludge, or mineral fertilizers) impacts on the heavy metal content of agricultural soils and hence on availability for transfer to waterbodies. Reducing manure use will reduce inputs of copper and zinc, for example, but the corresponding increase in mineral fertilizer required to maintain productivity may increase soil cadmium. Alternatively, changing farm management, for example, from arable to mixed farming, may reduce heavy metals at the source because of the overall reduction in fertilizers and manures used.

Minimizing the availability of heavy metals for transfer from the field can be achieved by increasing the ratio of grass to arable crop species or choosing particular arable crops to maximize plant uptake (e.g., metal removal from soil is much higher for sugar beet than for potatoes or wheat) (13). Alternatively, soil can be managed to minimize heavy metal transfer, through liming to raise pH levels and hence minimize leaching, or the application of soil conservation measures (such as minimum tillage and contour cultivation) to minimize heavy metal losses associated with soil erosion.

CONCLUSION

Water for food production, drinking, recreation, and ecology is an important but increasingly threatened global resource. The impacts of agricultural land use on the quality of water resources are numerous and include serious economic, health, and environmental implications from a wide range of specific pollutant threats. There are, however, a wide variety of measures available to minimize the sources, movement, and impacts of these agriculturally derived pollutants, most of which help to control pollution from more than one source. Control of water pollution within agricultural landscapes is, therefore, both necessary and achievable.

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CLASSIFICATION AND MAPPING OF AGRICULTURAL LAND FOR NATIONAL WATER-QUALITY ASSESSMENT

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INTRODUCTION

The U.S. Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) Program is designed to describe the status and trends in the quality of the nation's ground- and surface-water resources and to link the status and trends with an understanding of the natural and human factors that affect the quality of water. The program integrates information about water quality at a wide range of spatial scales, from local to national, and focuses on water-quality conditions that affect large areas of the nation or occur frequently within small areas.

The building blocks of the NAWQA Program are Study-Unit investigations, which will be conducted in 60 major hydrologic basins (Study Units) of the nation (Fig. 1). The Study-Unit investigations consist of intensive assessment activity for 3 years, followed by 6 years of less intensive monitoring, with the cycle repeated perennally (Leahy and others, 1990). Twenty Study Units will be in an intensive data-collection and analysis phase during each particular year, and the first complete cycle of intensive investigations of all 60 Study Units will be completed in 2002. The 60 NAWQA Study Units cover about one-half of the conterminous United States, encompass 60–70 percent of national water use and of the population

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served by public water supplies, and include diverse hydrologic systems that differ widely in the natural and human factors that affect water quality.

The distribution of Study Units ensures that the most important regional and national water-quality issues can be addressed by comparative studies among Study Units. NAWQA National Synthesis projects combine results of Study Unit Investigations with existing information from other programs and studies of the USGS and other agencies and researchers to produce regional and national-scale assessments for priority water-quality issues. The first water-quality issues to be focused on for National Synthesis are nutrients and pesticides.

Comparative studies among study units and national or regional aggregations of data for National Synthesis require consistent data on factors that affect the sources, behavior, and effects of contaminants and other factors that determine water-quality conditions. Natural features, such as geology, and factors related to human activities, such as land-use distribution, provide an environmental framework for assessing influences on water quality in different hydrologic systems. The emphasis of NAWQA on large-scale water-quality issues affected by human activities makes land-use characterization one of the most important aspects in study design and in the analysis of causes of water-quality conditions. Land-use characterization based on a nationally consistent classification system provides a basis for comparing the influences of human activities among Study Units. It also serves as a framework upon which to add many other types of county-based data on human activities, such as waste discharge or chemical use, that are associated with land use.

The purpose of this paper is to describe a system for classifying agricultural land for national water-quality assessment. The system focuses on classification of agricultural land within the 48 contiguous states. Alaska and Hawaii were included in basic data analysis but omitted from classification results because of their unique characteristics. Agricultural land use is one of the most important influences on water quality at national and regional scales, particularly in relation to the first NAWQA National Synthesis topics, nutrients and pesticides. Considered nationwide, agricultural land has

large areal extent, a high degree of land disturbance, and high use of agricultural chemicals and water. Individual areas of agricultural land, however, can vary widely in their characteristics. There is a great diversity of agricultural activities in the nation, which follow distinct regional patterns influenced by environmental setting and economics. Different mixes of agricultural activities characteristic of particular regions can have widely varying influences on water quality because of differences in management practices and natural environmental setting.

METAL TOLERANCE IN PLANTS: THE ROLES OF THIOL-CONTAINING PEPTIDES

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Chelation and compartmentalization are important mechanisms for metal tolerance in plants. Thiol-containing peptides, i.e., *glutathione* (GSH), *phytochelatins* (PCs), and *metallothioneins* (MTs), are metal ligands and play important roles in *metal tolerance* in plants. This article summarizes the present knowledge about the functions of these thiol-containing *peptides* in plant heavy metal tolerance, especially for Cd, As, and Cu. GSH participates in the amelioration of metal-induced oxidative stress. Metal-GSH complexes are the substrate for PC synthesis. Thus, enhanced GSH synthesis can increase metal tolerance in plants. PCs, a set of thiol-rich peptides with the general structure $(\gamma\text{-GluCys})_n\text{-Gly}$, play an essential role in constitutive Cd and tolerance, whereas PCs are not involved in adaptive Cd tolerance. In contrast, PCs are required for both constitutive and adaptive As tolerance in plants. PCs may play a role in constitutive Cu tolerance in some plants, but they are not involved in Ni or Zn tolerance. MTs may play an important role in constitutive Cu tolerance, and may be involved, to some extent, in Cd tolerance in plants.

INTRODUCTION

Heavy metals are those elements that have densities in excess of 5 g/cm^3 (1). About 40 elements fit this definition (in this paper, we consider metalloid arsenic (As) to be a heavy metal, because its properties are similar to those of other heavy metals). Heavy metals are often toxic to plants, although some heavy metals, i.e., Cu and Zn, are essential nutrients for plants and other organisms at low concentrations. Prolonged exposure to elevated concentrations can cause toxicity symptoms for both essential and nonessential heavy metals (e.g., Cd, Pb, Hg, As, etc.). The toxicity may result from deactivation of proteins via the binding of metals to sulfhydryl groups, competition for the active site between the heavy metal and an essential element resulting in deficiency effects (2,3), or production of free radicals and reactive species induced by heavy metals (3,4). Terrestrial plants are immobile organisms, and

metal ion uptake mechanisms are not very selective. As a consequence, plants will take up substantial amounts of toxic heavy metals from metalliferous soils. Plants must have developed strategies that enable them to tolerate the presence of toxic heavy metals. Possible mechanisms for metal tolerance in plant cells are: (a) metal binding to cell wall; (b) reduced transport across the cell membrane; (c) active efflux; (d) compartmentalization; and (e) chelation (5). Chelation and subsequent compartmentalization play important roles in metal tolerance in plants. A variety of compounds, such as organic acids, amino acids, glutathione (GSH), phytochelatins (PCs), and metallothioneins (MTs), have been identified as the chelators of heavy metals. Therefore, these ligands may be involved in metal tolerance in plants.

Metal tolerance mechanisms in plants are categorized as constitutive tolerance mechanisms that exist in normal plants and adaptive tolerance mechanisms that are present in metal-tolerant plants including hyperaccumulators (6). Constitutive metal tolerance mechanisms are complex and are controlled by multiple genes. Adaptive metal tolerance mechanisms, however, seem to involve relatively simple control mechanisms (7). Thiol-containing peptides glutathione (GSH), phytochelatins (PCs), and metallothioneins (MTs) are thought to play important roles in metal tolerance in plants (3,8–10). This article summarizes the present knowledge about the functions of these thiol-containing peptides in plant heavy metal tolerance, especially for Cd, As, and Cu.

GLUTATHIONE

Glutathione (GSH), the tripeptide γ -glutamylcysteinylglycine, is the major source of nonprotein thiols in most plant cells. GSH is synthesized from γ -glutamic acid, cysteine (Cys), and glycine (Gly) by a two-step ATP-dependent reaction. The first reaction synthesizes γ -glutamyl cysteine (γ -EC) from γ -glutamic acid and Cys by γ -glutamylcysteine synthetase (γ -EC synthase); the second step forms GSH from γ -EC and Gly catalyzing by GSH synthase. GSH plays an important role in protecting plants from oxidative stress induced by xenobiotics and some heavy metals, thus increasing metal resistance in plants (11). A substantial number of reports about oxidative stress induced by a variety of metals exist, e.g., Cd (12–14), Cu (4,15,16), Ni (17), Hg (18), and As (19). GSH is involved in the amelioration of metal-induced oxidative stress in plants. For example, a wild type of *Arabidopsis thaliana* exposed to Cu showed changes in GSH and glutathione disulfide (GSSG) levels, suggesting their participation in the amelioration of metal-induced oxidative stress (15). A decrease in GSH synthesis leads the hypersensitivity of plants to metals, which has been shown in *Arabidopsis* exposed to Cd (12), *Holcus lanatus* exposed to As (19), and *Silene cucubalus* exposed to Cu (4).

As metal-GSH complexes are the substrate of phytochelatin synthase (PCS) (9), synthesis of PCs may lead the depletion in the GSH pool. In the presence of metals, GSH-metal complex may limit the rate of PC synthesis (20). Therefore, an increase in GSH levels facilitates the increase in PC synthesis and the sequestration of

metals (13). It has been demonstrated that increases in GSH synthesis can improve metal tolerance in plants. For example, engineered Indian mustard, which overexpresses GSH, shows an increase in Cd tolerance and accumulation (20,21); elevated GSH concentrations are also involved in conferring tolerance to Ni-induced oxidative stress in *Thlaspi* Ni hyperaccumulators (17).

Vacuolar compartmentalization is an important mechanism for heavy metal tolerance in bacteria and plants. Some metals, i.e., Cd and As, need PCs to mediate the vacuolar compartmentalization in bacteria and plants (22–25). Recent studies indicate that GSH can also mediate vacuolar compartmentalization of metals, e.g., Cd, As, and Hg, in bacteria (26–28). In yeast *Saccharomyces cerevisiae*, which lacks PCS, a Cd-GSH complex can be directly transported to vacuoles by YCF1, one of ATP-binding cassette (ABC) transporter proteins located in tonoplasm (26,29–31). YCF1 was also found to transport As-GSH and Hg-GSH complexes to vacuoles in bacteria (27,28). Recently, BPT1, another ABC transporter in vacuoles, was found in yeast. Similar to YCF1, BPT1 also transports GSH conjugates including the Cd-GSH complex (32). However, this mechanism of metal tolerance involving active sequestration of the metal-GSH complexes into vacuoles by ABC transporters has not yet been reported in plants.

PHYTOCHELATINS

Structures

PCs are a set of thiol-rich peptides with the general structure $(\gamma\text{-GluCys})_n\text{-Gly}$ ($n = 2$ to 11), which are synthesized from GSH catalyzed by PCS (33–36). They were first identified in the yeast *Schizosaccharomyces pombe* and termed Cadystins (37,38), and isolated and characterized by Grill et al. (39). PCs have been found in all plants studied to date and some microorganisms (40). They are also found in animals, i.e., nematode *Caenorhabditis elegans* (41). Interestingly, some iso-phytochelatins (iso-PCs) are identified in some plants under heavy metal exposure (1). Grill et al. found that a few members of the *Fabales* produced homoPCs under Cd exposure (42). HomoPCs are synthesized from homoglutathione, in which β -Ala replaces Gly as a terminal amino acid (42). Klapheck et al. found that hydroxymethyl-PCs [$(\gamma\text{-GluCys})_n\text{-Ser}$] were induced by Cd in several species of the family Poaceae (43). They are thought to be formed from hydroxymethyl-glutathione [$(\gamma\text{-GluCys})_n\text{-Ser}$]. Meuwly et al. discovered $\gamma\text{-Glu-Cys-Glu}$, another homologue of GSH. It is not surprising that another iso-PC, $(\gamma\text{-GluCys})_n\text{-Glu}$ ($n = 2 - 3$), was induced by Cd in maize seedlings (44). Recently, iso-PCs $(\gamma\text{-GluCys})_n\text{-Gln}$ ($n = 3 - 4$) were isolated from the hairy roots of horseradish exposed to Cd (45). In addition, some PC-related compounds, i.e., desglycine PCs [$(\gamma\text{-GluCys})_n$] (46,47), Cys- $(\gamma\text{-GluCys})_n\text{-Gly}$ ($n = 1-5$), and Cys- $(\gamma\text{-GluCys})_n$ (47), were identified in maize.

Biosynthesis

PCs are not primary gene products. They are synthesized from GSH by a specific γ -glutamylcysteine

dipeptidyl transpeptidase (EC 2.3.2.15)—PCS in the presence of metal ions (48). This enzyme transfers γ -Glu-Cys from GSH or PC to an acceptor molecule (i.e., GSH and PC) (49). PC synthesis reaction can be rapidly induced *in vivo* by exposure to a variety of metal cations (Cd^{2+} , Pb^{2+} , Zn^{2+} , Sb^{3+} , Ag^+ , Hg^{2+} , Cu^+ , Sn^{2+} , Au^{3+} , Bi^{3+}) and metalloid anions [arsenate (As^{V}) and arsenite (As^{III})] (1,3,10,37,50,51). The reaction is self-regulated. Once the free metal ions are chelated by the reaction products PCs, the reaction terminates (48,49,52). PCS activity was identified for the first time in cell-free extracts from suspension cultures of *Silene cucubalis* by Grill et al. (48), and similar PCS activities have been detected in pea (52), tomato (49), and *Arabidopsis* (53). In 1999, genes encoding PCS were first cloned and characterized in *Arabidopsis* (*AtPCS1*), *S. pombe* (*SpPCS*), and wheat (*TaPCS1*) (54–56). Homologous genes encoding PCS have been identified in other species, i.e., *C. elegans* (*CePCS*) (57), *Arabidopsis* (*AtPCS2*) (55,58), soybean *Glycine max* (*GmhPCS1*) (59), and *Brassica juncea* (Indian mustard) (*BjPCS1*) (60).

PCS is expressed constitutively, and the levels of this enzyme in cell cultures or intact plants are generally unaffected by exposure to metal ions (61). Analysis of the expression of *AtPCS1* indicated that levels of mRNA were not affected by exposure of plants to Cd (54,55). However, RT-PCR analysis of *TaPCS1* expression in wheat roots indicated induction of mRNA under Cd exposure (56). In addition, a recent study showed *AtPCS1* of *Arabidopsis thaliana* was regulated at transcriptional level during the early stages of plant development (62). Therefore, in some species or specific stages of plant development, PCS activity may be regulated both at transcriptional and posttranslational levels.

Earlier studies have shown that PCS activity requires the binding of free metal ions to the enzyme (48,49,52,63,64). However, a recent study has shown that metal binding per se is not responsible for the activation of PCS, but glutathion-like peptides containing blocked thiol groups are required for the enzyme activity (9). A similar conclusion was reached by Oven et al. (59). Therefore, the real substrates of PCS are metal-thiolate complexes.

Roles of PCs in Metal Tolerance

PCs are suggested to play a role in metal detoxification and homeostasis (1,37). However, no direct evidence that PCs are involved in homeostasis currently exists (51). It has been demonstrated that PCs play an essential role in constitutive Cd and As tolerance in plants. In addition, PCs may be required for adaptive enhanced As tolerance. Some indications exist that PCs may play a role in Ag, Cu, and Hg tolerance (55,56). However, the role of PCs in Cu tolerance in plants has been debated (54–56,65). A paucity of information exists about the roles of PCs for Ag and Hg tolerance in plants. Zn and Pb are weak activators for PC synthesis (48,50), and genetic and biochemical evidence has shown that PCs are not involved in Zn tolerance in plants (53–55,65–68). No evidence exists that PCs are involved in Pb tolerance in plants. Ni cannot induce PC

synthesis (48,50), thus PCs are not involved in Ni tolerance in plants.

Cadmium. Cd is a strong activator for PC synthesis, and PCs and Cd can form detectable complexes *in vivo* (48,50). PCs play an essential role in normal constitutive Cd tolerance in plants (3,10,37,51,64,69,70). This conclusion is supported by a variety of biochemical and genetic evidence. *In vitro*, PCs have been shown to substantially decrease the toxicity of Cd to metal-sensitive plant enzymes, such as alcohol dehydrogenase, glyceraldehyde 3-phosphate dehydrogenase, nitrate reductase, ribulose 1,5-diphosphate carboxylase, and urase (71). In normal *S. vulgaris*, treatment with buthionine sulfoximine (BSO), which inhibits the synthesis of the PC precursor γ -EC, resulted in hypersensitivity to Cd (72). The most convincing evidence comes from genetic studies. Cd-sensitive *cad1* mutants of *Arabidopsis* are deficient in PCS activity and do not form Cd-PC complexes. Consequently, these mutants are sensitive to Cd (67,68). Expression of cDNA clones of *AtPCS1* from *Arabidopsis* and a similar gene *TaPCS1* from wheat in *S. cerevisiae* resulted in a dramatic increase in Cd tolerance (55,56). The mutants of *Arabidopsis* and *S. pombe* that lack PCS genes exhibited hypersensitivity to Cd and were unable to synthesize PCs (54,56). Azuki bean cells lacking PCS activity were also hypersensitive to Cd (73). All of this evidence strongly suggests that PCs play an essential role in constitutive Cd tolerance.

In addition to formation of Cd-PC complexes in cytoplasm, sequestration of Cd-PC complexes to vacuoles is also required for Cd tolerance (22,74). In vacuoles, the complexes are stabilized by incorporation of acid-labile sulfide (75). In yeast *S. pombe*, vacuolar sequestration process is mediated by HMT1 (22). HMT1, a member of ABC-type transport proteins located in tonoplasm, can transport PCs and Cd-PC complexes to vacuoles (76). Vacuolar compartmentalization of Cd-PC complexes has also been observed in plants, such as tobacco (75,77), and oat roots (23). However, plant genes encoding Cd-complex transporter have not been characterized yet (40).

Although PCs are essential for normal constitutive Cd tolerance, PCs are not involved in adaptive enhanced Cd tolerance in Cd-tolerant plants. In studies that compared Cd-tolerant plants with normal *Silene vulgaris*, De Knecht et al. found that Cd-tolerant plants exhibited a much lower PC synthesis and a lower rate of longer PC synthesis under Cd exposure than normal plants, and their Cd tolerance was not reduced by BSO treatment (72,78). These observations suggest that the enhanced Cd tolerance in Cd-tolerant *S. vulgaris* is not because of differential production of PCs and synthesis of longer chain PCs, although PCs are essential for constitutive Cd tolerance in normal *S. vulgaris* (72,78). Similar results were obtained in other studies of *S. vulgaris* (65,79). In Zn/Cd hyperaccumulator *Thlaspi caerulescens*, PC synthesis under Cd exposure was lower than in nonhyperaccumulators *Thlaspi arvense* (80), and BSO treatment did not increase sensitivity to Cd (65). These results suggest that PCs are not involved in adaptive enhanced Cd tolerance in this plant. The possible

mechanism for the adaptive enhanced Cd tolerance in these Cd-tolerant plants is PC-independent vacuolar sequestration (65). However, this process remains to be elucidated.

Arsenic. Similar to Cd, As^{III} can induce high levels of PC synthesis in plants (50,81). However, it is difficult to detect As-PC complexes *in vivo*, although formation of As^{III}-tris-thiolate complexes is predicted on a theoretical basis (50). With improved experimental conditions, Schmoger et al. demonstrated the formation of As^{III}-PC complexes in plants by size exclusion chromatography and electrospray ionization-mass spectrometry (24). Constitutive As tolerance in plants requires the involvement of PCs. This process may include reduction of As^V to As^{III}, formation of As^{III}-PC complexes, and sequestration of the complexes into vacuoles (65,81,82). Results from various studies support the role of PCs in constitutive As detoxification in normal plants. For example, BSO-treated cell cultures of tobacco (83,84) and *Rauwolfia serpentina* (83,84), *Holcus lanatus* (6,82,85), and *Cytisus Striatus* (6,82,85) were found to be hypersensitive to As. The strongest evidence for the role of PCs in As detoxification comes from genetic studies of *Arabidopsis* and *S. pombe* (54,55). PCS-deficient mutants of *Arabidopsis* and *S. pombe* are highly sensitive to Cd and As (54). Conversely, the expression of *AtPCS1* from *Arabidopsis* in yeast *S. cerevisiae* resulted in enhanced resistance to As (55). This evidence strongly supports the argument that PCs are essential for constitutive As tolerance in plants.

Unlike enhanced Cd tolerance in Cd-tolerant plants, enhanced As tolerance in As-tolerant nonhyperaccumulator plants may require PCs. In As-tolerant *H. lanatus* and *C. Striatus*, suppression of phosphate/As^V uptake via an altered high-affinity phosphate/As^V uptake system is the adapted As tolerance mechanism that is mainly responsible for the enhanced As tolerance (6,19,82,86). However, PCs are also involved in the enhanced As tolerance in these As-tolerant plants via synthesis of higher levels and/or longer chain of PCs, because BSO treatment can cause these As-tolerant plants to be more sensitive to As (6,82,85). These observations suggest that constitutive production of PCs may be involved in the adaptive As tolerance in these plants. In As hyperaccumulator *Pteris vittata*, the major mechanism responsible for As hyper-tolerance is not related to restriction of As uptake, but may be related to enhanced vacuolar compartmentation of As (87). PCs were suggested to play no role in As hyper-tolerance in *P. vittata* (88). However, considering that PCs are required for enhanced As tolerance in the As-tolerant nonhyperaccumulators, PCs may play, to some extent, a role in As hypertolerance in this hyperaccumulator, which is supplementary to the adaptive As tolerance mechanisms (87).

Copper. Cu is an essential micronutrient for plants. The mechanisms for plant Cu tolerance have not been elucidated. Cu ion can induce PC synthesis, and formation of Cu⁺-PC complexes are observed *in vitro* and *in vivo* (89,90). The role of PC in Cu tolerance in plants has been debated. Some indications exist that PCs may, to

some extent, be involved in constitutive Cu tolerance. For instance, *S. pombe* cells, for which the PCS gene expression has been disrupted, exhibit hypersensitivity to Cd and Cu (56); PCS-deficient *Arabidopsis* mutant *cad1-3* is slightly sensitive to Cu, Hg, and Ag (54,91). However, the evidence is not conclusive. Another genetic study showed that PCS-deficient *S. pombe* was insensitive to Cu, Zn, Hg, Se, or Ni ions (54). *Arabidopsis* exposure to concentrations of Cu ion did not produce Cu-PC complexes, but a change in glutathione and glutathione disulfide contents was observed (15). In normal *S. vulgaris*, BSO treatment had no effect on the plant sensitivity to Cu, indicating that PCs are not essential for constitutive Cu tolerance (65).

PCs are not involved in adaptive enhanced Cu tolerance in Cu-tolerant plants. Differential Cu tolerance in Cu-tolerant and normal *S. vulgaris* is not because of differential PC production (92). The sensitivity to Cu is unaffected by BSO treatment both in Cu-tolerant and normal *S. vulgaris*, suggesting that PCs are not involved in adaptive Cu tolerance (65). A similar conclusion was obtained in Cu-tolerant *S. cucubalus* (4).

METALLOTHIONEINS (MTs)

MTs are a class of Cys-rich polypeptides (Mr: 6000–7000) that can bind metals via thiol groups of their Cys residues (93). MTs were first reported as Cd binding protein in horse kidney (94). MTs have been found in animals, plants, eukaryotic microorganisms, and some prokaryotes (40). MTs contain characteristic motifs: Cys-Cys, Cys-X-Cys, and Cys-X-X-Cys (5). MTs are divided into two classes (95). Mammalian MTs containing 20 highly conserved Cys residues are categorized as Class I (96), whereas MTs from plants, fungi, and invertebrate animals are grouped in Class II (97). According to the arrangement of Cys residues, plant MTs are further classified into four types (40,95). MTs are gene-encoded polypeptides, and their syntheses are transcriptionally activated by several factors such as hormones, cytotoxic agents, and metals, including Cd, Zn, Hg, Cu, Au, Ag, Co, Ni, and Bi metal ions (5). A wide variety of regulatory factors and expression in many organisms suggest that MTs have important functions (40). Although MTs have been studied for more than 40 years, their function has not been completely understood in animals (3,98). They may play a role in protection against metal toxicity, function as antioxidants, and are thought to be involved in plasma membrane repair (3). The only confirmed function of MTs in animals is detoxification of Cd and Zn (96,98).

MTs have been less studied in plants than in animals. The functions of MTs in plants also remain to be elucidated. Some indications exist that MTs may play a role in Cu and probably Cd tolerance in plants. Cu tolerance in *Arabidopsis* depends on the degree of Cu-induced expression of type 2 MT genes (99,100). When MT1a and MT2a genes from *Arabidopsis* were expressed in a yeast strain lacking one of its endogenous MT genes (*cup1*), the MT genes conferred the yeast a high level of tolerance to Cu and a moderate level of tolerance to Cd (101,102). Transgenic tobacco plant with expression

of yeast MT gene can accumulate significantly higher quantities of Cu than control plants (103). When an MT2b-like gene (*SvMT2b*) from a highly Cu-tolerant *S. vulgaris* was expressed in yeast, this gene restored cadmium and copper tolerance in different hypersensitive strains (104). When an MT2-like gene (*tyMT*) from *Typha latifolia* was expressed in Cu-sensitive MT-deficient strain of yeast, it restored the yeast's metal tolerance (105). Transgenic *Arabidopsis*, with expression of the *tyMT* gene, showed an increased tolerance to heavy metals Cu and Cd (105). Results from these studies suggest that MTs play an essential role in constitutive Cu tolerance in these plants.

However, other studies do not support this conclusion. For instance, In Indian mustard, Cu treatment caused a strong increase in γ -ECS mRNA, GSH, and PCs in roots and shoots, whereas MT2 mRNA was decreased in response to the CuSO₄ treatments (106,107); In *Vicia faba*, MT1 and MT2 transcript levels are not significantly affected by treatment with Cu, Zn, or Cd (108,109). The observed decrease or no change in MT expression under Cu exposure may be caused by increased levels of PC synthesis, suggesting that Cu quenched by PCs may be favored over sequestration by MTs in Indian mustard and *Vicia faba* (106,109). As mentioned above, genetic evidence has shown that PCs may play a role in constitutive Cu tolerance in some plants (55,56), and constitutive metal tolerance mechanisms are complicated and controlled by multiple genes (7). Therefore, both PCs and MTs may be involved in constitutive Cu tolerance in plants. For some reasons, the PC-dependent mechanism appears to dominate in some plants, whereas the MT-dependent mechanism is favored in others.

MTs may also be involved in adaptive enhanced Cu tolerance in plants. Cu-tolerant *S. vulgaris* was found to have significantly higher transcript levels of type 2b MT gene than normal *S. vulgaris* (104). The increase in the MT gene expression suggests that constitutive production of MTs also contribute to the enhanced Cu tolerance, although an adaptive mechanism is mainly responsible for the enhanced Cu tolerance in *S. vulgaris* (104). Restriction of Cu uptake may be the adaptive mechanism for the enhanced Cu tolerance in plants. By comparison of normal to Cu-tolerant *Silene cucubalus*, De Vos et al. suggested that the adaptive Cu tolerance in this plant does not depend on the production of PCs but is related to the plant's ability to prevent GSH depletion resulting from Cu-induced PC synthesis, e.g., by restriction Cu uptake (4). A recent study has shown that efflux of Cu across the root plasma membrane plays a role in the Cu tolerance in Cu-tolerant *S. vulgaris* (110).

In animals, MTs play an essential role in Cd detoxification. In plants, this role is mainly occupied by PCs (40). However, MT expression can enhance Cd tolerance in plants. For instance, expression of mouse MT1 gene and *Nicotiana glutinosa* MT gene confers Cd resistance in transgenic tobacco plants (111,112), suggesting that MT may play a role, to some extent, for Cd tolerance in plants. An increase in Cd tolerance was also found in transgenic *Arabidopsis* with expression of an MT gene (105). The roles of MTs in Cd tolerance in plants need further investigation and confirmation.

CONCLUSIONS

GSH, PCs, and MTs play important roles in metal tolerance in plants. GSH participates in the amelioration of metal-induced oxidative stress, and metal-GSH complexes are the substrate for PC synthesis. Thus, enhanced GSH synthesis can increase metal tolerance in plants. PCs play an essential role in constitutive Cd tolerance, whereas PCs are not involved in adaptive Cd tolerance. In contrast, PCs are required for both constitutive and adaptive As tolerance in plants. PCs may play a role in constitutive Cu tolerance in some plants, but they are not involved in Ni or Zn tolerance. MTs may play an important role in constitutive Cu tolerance, and may be involved, to some extent, in Cd tolerance in plants.

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MICROIRRIGATION

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Microirrigation has been defined in the Standards of the American Society of Agricultural Engineers (ASAE) as “frequent application of small quantities of water on or below the soil surface in form of drops, tiny streams or miniature spray.” In microirrigation systems, water is applied through emitters or applicators placed along a water delivery line (lateral line) adjacent to the plant row (1). Microirrigation includes all methods of frequent water application, in small flow rates, on or below the soil surface. The terms drip, trickle, and spray irrigation, common since the 1960s, have been supplanted by the term

“microirrigation,” which is now commonly used throughout the industry. Microirrigation is sometimes called “localized irrigation” to emphasize that only a targeted part of the soil volume is wetted during water application. The other name that is frequently and incorrectly used is “low volume irrigation.” In reality, a microirrigation system is characterized by low flow rates that are expressed in terms of water volume per time (L/h, m³/h), but the total volume (liters or cubic meters) delivered can be large.

Microirrigation systems are designed to transport water from a source through a delivery network of pipes and various types of emitters to a crop. The general goal of a microirrigation system is to provide irrigation water uniformly and efficiently to a crop to help meet the water needs of plants and to maintain a favorable root zone water balance. Ideally, the volume of water is applied directly to the root zone in quantities that approach the consumptive use of the plants. Additional goals of microirrigation can be to increase fruit or fiber production, to maintain crop visual quality, to protect plants and/or fruits and flowers from extreme temperature conditions (hot or cold), to apply crop or soil chemicals (fertilizers, pesticides, etc.), or to dispose of wastewater, which can be an important water and nutrient resource.

Microirrigation is usually characterized by the following features: (1) low water application rates, (2) high frequency of water application, (3) water applied near or into the root zone, (4) low pressure delivery system, and (5) delivery of fertilizers and other agricultural chemicals with irrigation water (fertigation and chemigation).

In microirrigation systems, water is distributed using an extensive hydraulic pipe network (Fig. 1) that conveys water from its source to the plant. Outflow from the irrigation system occurs through emitters placed along the water delivery (lateral) pipes in the form of droplets, tiny streams, or miniature sprays. The emitters can be placed either on or below the soil surface. Microirrigation systems are classified by the type of emitters used in the system. These are drip, bubbler, spray jet (or microspray), and microsprinkler. The system can also be subdivided into the point source emitters and line source emitters (Fig. 1). Point source emitters are spaced sufficiently far apart to create individual discrete water patterns and they are used for trees or widely spaced shrubs. Line source emitters are used for row crop production. The emitters are placed along the line closely enough to create a continuous wetted band along the crop row. In addition, line source (drip) systems can apply water above the ground or below the ground. Subsurface drip irrigation (SDI) systems are becoming more popular due to potentially high efficiency.

A typical flow rate of a point source emitter is less than 12 L/h. The flow rate of a line source emitter is given per length of lateral and is also less than 12 L/h per meter. Bubblers emit much larger volumes of water, but usually less than 225 L/h, where the microsprinklers and jet sprays are limited to 175 L/h.

The emitter is designed to reduce the operating pressure from the supply line, and a small volume of water is discharged at the emission point at low pressure. Emitters can vary from sophisticated, constant

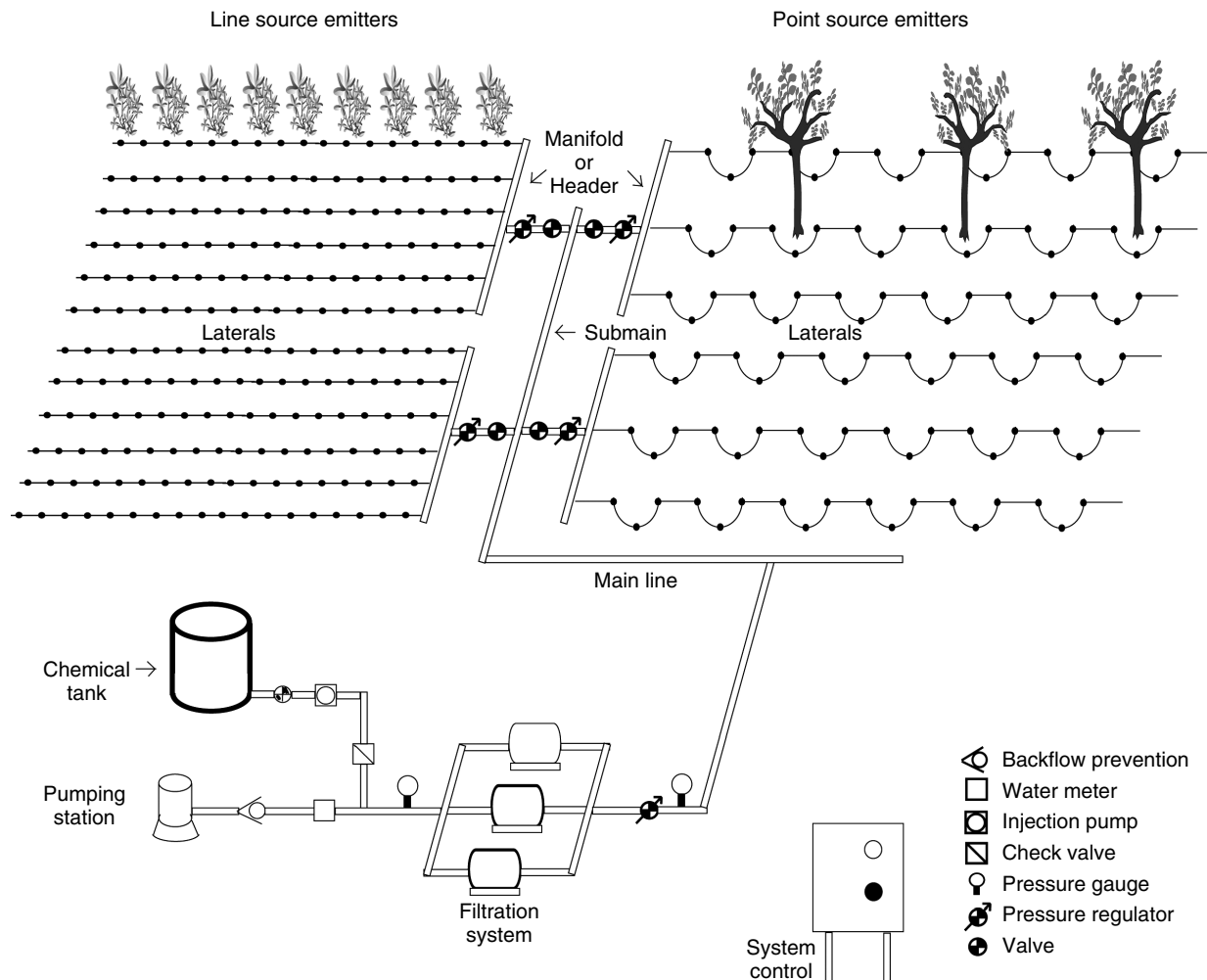


Figure 1. An example of a basic microirrigation system.

flow rate at variable pressure types of devices (pressure-compensating emitters) to very small, simple orifices. A large number of different types of emitters have been developed in attempts to find a perfect device. The main objective is to assure uniformity of water distribution. It is essential that the discharge from the emitter is uniform and that it not change significantly with small pressure variations in the system. At the same time, the emitter should be constructed in such a way that it does not clog very easily. The cost and the size are also important. The emitters presently available on the market can be classified into five distinct categories: (1) long path emitters, (2) short path orifice emitters, (3) vortex emitters, (4) pressure-compensating emitters, and (5) porous pipe or tube emitters (line source drippers).

Emitters are placed along laterals, the final water delivery lines, designed for uniform water distribution. The lateral line is generally constructed of flexible polyvinyl chloride (PVC) or polyethylene hose (PE). It is often placed above the ground, but it can be buried. For row crops, in the line source type of microirrigation system, a lateral line combines the function of the line and the emitter. These include laterals constructed from porous

pipe, twin-bore PE pipe, or PE pipe provided with evenly spaced, built-in emitters. Each manufacturer of line source emitters supplies data with allowable length of laterals and recommended pressure for particular line source systems. Recommendations are based on the hydraulic characteristics of the product. The emitters, which may be connected to the lateral using a variety of connectors during installation of the irrigation system, are usually placed at predetermined positions, for example, at the base of the irrigated tree.

Lateral lines are attached to manifolds or submains. The manifolds and submains, which distribute the water to the specific parts of the field, are usually constructed from flexible smooth-walled, noncollapsible, black PE or flexible PVC pipe that can be left on the soil surface or buried. If rigid (usually white or purple) PVC pipe is used for the lateral line, it should be buried beneath the soil surface for protection against sunlight damage and prevention of algae growth in the line. The controls for adjustment of flow rate and pressure are usually located in the submain or manifold line along with valves and timing devices for the separate parts of the field. Water is delivered to the field by the main line. The main line

is usually constructed from white PVC pipe that is buried beneath the soil surface for protection against harmful sunlight. Purple color is required for all PVC components in the system using reclaimed wastewater. The pipe should be properly rated for the particular application and able to withstand the design pressure in the system.

A main control station, often called the “control head,” is usually located close to the water supply. A typical control station includes the pump, a backflow prevention system, a chemical injection system for fertilizers, chlorine, or other chemicals, and a combination of different filters. A main line valve and flowmeter are also included in the control head. Microirrigation systems can be controlled manually or automatically. Automatic control can be electromechanical (clock) or electronic (computer). The controller is often located next to other components of the control station. It can control the main valve, chemical injection, backflushing of filters, solenoid valves, and other controls located at remote locations in the irrigation system. Depending on the system, all or some of these components can be automated.

In microirrigation systems, only a portion of the total field area is wetted. The goal is to only wet the root zone of the crop of interest. Water flowing from the emitter is distributed in the soil by gravity and capillary forces creating the contour lines presented in Fig. 2, often referred to as “onion” patterns. The exact shape of the wetted volume and moisture distribution will depend on the soil texture, initial soil moisture, and, to some degree, the rate of water application. In the line source type of microirrigation tube that is frequently used for row crops such as vegetables, where the emitters are spaced very closely, individual “onion” patterns connect, creating a continuous moisture zone along the row.

Microirrigation systems have many potential advantages compared to other irrigation methods. Most of them are related to the low rates of the system. It can be argued that some of these benefits are not unique to a microirrigation system. However, certain combinations of these

advantages are responsible for uniqueness of microirrigation in contrast to other systems.

WATER SAVINGS

Irrigation water requirements can be smaller with microirrigation when compared with other irrigation methods since a smaller portion of the soil volume is wetted, there is decreased evaporation from the soil surface, and there is reduction or elimination of the runoff. The losses due to evaporation from the soil are significantly reduced compared with other irrigation systems because only a small surface area under the plant is wetted and it is usually well shaded by the foliage. A microirrigation system allows for a high level of water control application and water can be applied only when needed in precisely controlled amounts. Deep percolation can be minimized or avoided.

SMALLER FLOW RATES

Since the rate of water application (L/S) in microirrigation systems is significantly lower than in other systems, smaller sources of water can be used for irrigation of the same acreage. The delivery pipes, the pump, and other components of the system can be smaller and therefore more economical. The systems operate under low pressure (35–200 kPa) and require less energy for pumping than high pressure irrigation systems.

APPLICATION OF CHEMICALS

Microirrigation systems allow for a high level of control of chemical applications. The plants can be supplied with the exact amount of fertilizer required at a given time. They are applied directly to the root zone and reduction in the total amount of fertilizer used is possible. There is also an advantage to the frequent application of fertilizers

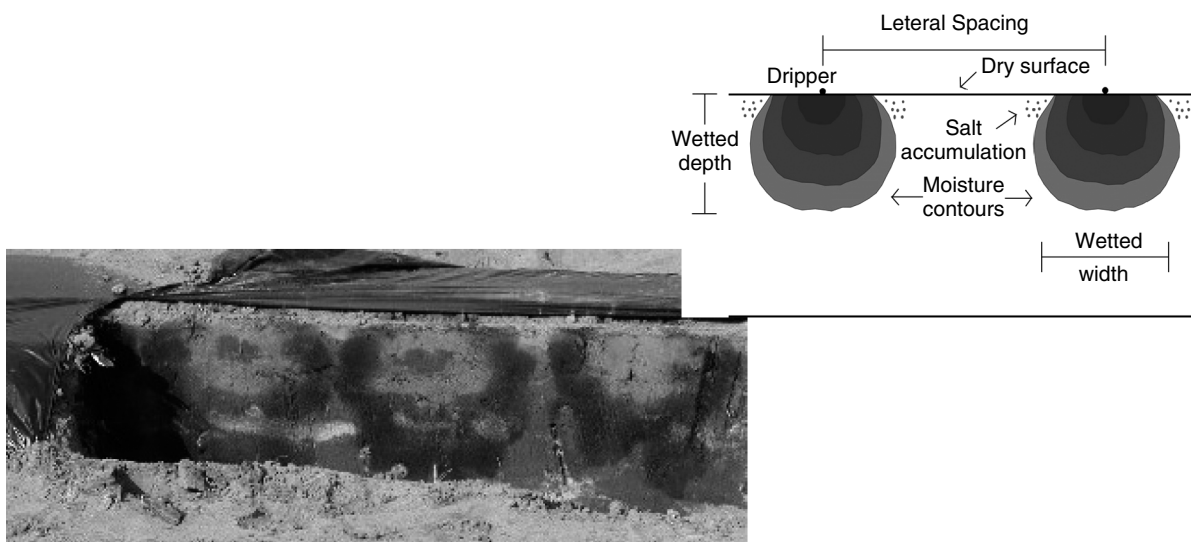


Figure 2. Wetted pattern under drip irrigation (line source type of emitter) in a sandy soil.

through the system (fertigation) in humid climates. In case of rain, only a small portion of recently applied fertilizer will be washed out and it can easily be replaced through the irrigation system, reducing the potential environmental hazard due to nutrient leaching from the production system. Fertigation is more economical, provides better distribution of nutrients throughout the season, and reduces groundwater pollution due to the high concentration of chemicals that could ordinarily move with deep percolated water. Other chemicals, such as herbicides, insecticides, fungicides, nematicides, growth regulators, and carbon dioxide can also be efficiently applied through microirrigation systems to improve crop production. This process of chemical application is called chemigation.

WATER SOURCES WITH HIGH SALT CONTENT

Water with relatively high salt content can be used in microirrigation systems. For optimum plant growth a certain range of total water potential in the root zone must be maintained. The potential defines how difficult it is for a plant to extract water from the soil. Large negative numbers are characteristic of very dry soils with low total water potentials, while potentials near zero reflect soils near saturation. The total water potential in the root zone is a sum of the matrix potential and osmotic potential. Since matrix potential is close to zero under well-managed microirrigation (high moisture content), the osmotic potential component can be a relatively large negative value, indicating high salt content, without a harmful effect on plant growth. This is not true for other irrigation systems.

IMPROVED QUALITY OF THE CROP

Microirrigated plants are supplied very frequently with small amounts of water and the stress due to the moisture fluctuation in the root zone is reduced to the minimum, often resulting in larger and better quality yield. In arid climates, or during dry seasons, proper water management can control the harvest timing.

ADAPTATION TO ANY TOPOGRAPHY

Microirrigation systems can operate efficiently on hilly terrain if appropriately designed and managed. A well-managed microirrigation system will minimize runoff even on hilly terrain.

ADDITIONAL ADVANTAGES OF MICROIRRIGATION SYSTEMS

During dry seasons or in arid climates, disease and insect damage can be reduced under the microirrigation system since the foliage of the plant is not wetted. With a small portion of soil surface being watered, field operations can be continued during irrigation. Water distribution is not affected by the wind in drip irrigation. However, wind

can have some effect on spray jet patterns. Germination and water uptake by the weeds between the rows can also be significantly reduced since these areas remain dry. Microirrigation systems can also be extensively automated, decreasing labor and operating costs.

To operate satisfactorily, a microirrigation system has to be correctly designed and managed to account for the physical properties of soil, quality of irrigation water, and water requirements of the grown plants. Usually, microirrigation requires a higher management level than other irrigation systems. With all the advantages listed above, a microirrigation system is not a system without problems.

CLOGGING

One of the biggest problems encountered under microirrigation is the clogging of the emitters. The small openings can easily be clogged by soil particles, organic matter, bacterial slime, algae, or chemical precipitates. Microirrigation systems require very good filtration (most often recommended is 200 mesh screen) even with a high quality water supply.

MOISTURE DISTRIBUTION

Moisture distribution depends largely on the soil type being irrigated by the microirrigation system. In some soils, for example, deep sands, very little lateral water movement (low capillary forces) can create many problems. Under these conditions it is difficult to wet a significant portion of the root zone since most of the water moves downward due to gravity. It is also more difficult to manage the irrigation without deep percolation since only a small amount of water can be stored in the wetted volume desired. Increasing the number of emitters per plant may improve water distribution in the soil. As a result, coarse sands require much closer spacing of emitters than fine soils. In general, for any soil, the number of emitters and their spacing must be based on the geometry of wetted soil volume. It is important to realize that the microirrigation system wets only a limited portion of the potential soil root volume. Most of the plants can perform very well under these conditions. However, there is a minimum volume of roots that have to be wetted or a reduction in yield will result.

SALT BUILDUP

Microirrigation systems can use saline water. However, a problem may occur from salts accumulating at the edges of the wetted zone during prolonged dry periods. Light rain can wash these salts into the root zone and cause injury to the plants. In arid climates, where the rainfall is less than 10 in./yr, an additional irrigation system (sprinkler or surface) may be necessary to leach accumulated salts from the soil. In areas with heavy rainfall the salts will be washed out of the root zone before significant accumulation occurs.

Table 1. Extent of Microirrigation (in hectares) in Selected Countries and the World During the Period 1981–2000

Country	1981	1986	1991	2000
United States	185,300	392,000	606,000	1,050,000
India	20	0	55,000	260,000
Australia ^a	20,050	58,758	147,011	258,000
Spain	0	112,500	160,000	230,000
South Africa	44,000	102,250	144,000	220,000
Israel	81,700	126,810	104,302	161,000
France	22,000	0	50,953	140,000
Mexico ^a	2,000	12,684	60,000	105,000
Egypt	0	68,450	68,450	104,000
Japan ^a	0	1,400	57,098	100,000
Italy	10,300	21,700	78,600	80,000
Thailand ^a	0	3,660	45,150	72,000
Colombia ^a	0	0	29,500	52,000
Jordan	1,020	12,000	12,000	38,300
Brazil ^a	2,000	20,150	20,150	35,000
China	8,040	10,000	19,000	34,000
Cyprus	6,000	10,000	25,000	25,000
Portugal	0	23,565	23,565	25,000
Chinese Taipei ^a	0	10,005	10,005	18,000
Morocco ^a	3,600	5,825	9,766	17,000
Other ^a	50,560	38,821	100,737	177,000
Total world	<u>436,590</u>	<u>1,030,578</u>	<u>1,826,287</u>	<u>3,201,300</u>

^a Areas for these countries adjusted from 1991 figures according to the average percentage increase of other countries that had provided updated information to ICID.

Source: Adapted from Reference 2.

INITIAL COST

The initial investment and maintenance cost for a microirrigation system may be higher than for some other irrigation methods. Filters, chemical injectors, and possible automation components add to the cost of a microirrigation system. Actual costs will vary considerably depending on the selection of a particular microsystem, required filtration equipment, water quality, water treatment, and selection of automation equipment.

ADDITIONAL DRAWBACKS

Rodents and insects can create additional maintenance problems by chewing holes in the plastic. In addition, some components of the system can easily be damaged by persons unaware of their locations. A microirrigation system does not provide significant frost protection when compared to sprinkler irrigation; however, it is commonly used for this purpose in citrus production where it can provide some protection.

CURRENT IRRIGATED AREA

In 1977, the Food and Agriculture Organization of the United Nations (FAO) estimated that the total global irrigated area was 223 million ha. By 1996 this estimate increased to about 262 million ha. The 2000 irrigation survey conducted by the *Irrigation Journal* (2001) listed 25.5 million ha irrigated in the United States. Approximately 12.7 million ha (49.9%) were irrigated by

sprinkler irrigation, 11.5 million ha (45.1%) by gravity irrigation, and 1.3 million ha (4.9%) by microirrigation.

Results of the surveys conducted by the International Commission on Irrigation and Drainage (ICID), summarized (Table 1) by Reinders (2), indicate that microirrigation has increased from 0.4 to 3.2 million ha between 1981 and 2000. In the United States, California, Florida, Washington, Texas, Hawaii, Georgia, and Michigan account for approximately 91% of the microirrigated land area. It is a small fraction of the total irrigated area; however, many of the high value crops that require intensive production practices are grown under microirrigation and the area of marginal lands that are microirrigated is increasing.

Almost all crops can be suitable for microirrigation; however, due to the high cost, microirrigation is primarily used on high value perennial crops, tree and vine crops, fruits, vegetables, and ornamentals. There is growing interest in applying microirrigation to lower valued field crops, such as cotton and corn or sugar cane, through the use of multiyear subsurface drip irrigation (SDI). Application of microirrigation for landscaping, greenhouses, and nurseries has also increased tremendously. Large containerized nursery plants are typically irrigated with microirrigation. In addition, an increase of microirrigation in residential properties and highway roadsides often prompted by water restriction and/or high cost can be observed in many areas of the United States.

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MICROIRRIGATION: AN APPROACH TO EFFICIENT IRRIGATION

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Microirrigation, earlier known as Drip or Trickle irrigation is a highly efficient method of water application. It developed and expanded globally at a fast rate in the last two decades in more than 35 countries with maximum coverage in the United States. This is most appropriately applicable to widely spaced crops such as fruits, vegetables and field crops. Its different variations consist of surface trickle, bubbler, microsprinkler, spray, mechanical move, pulse, subsurface drip, all of which are covered under a single term microirrigation. In the present article, its advantages, components, system design, fertigation, automation and other aspects has been described. Also the extent of application to different crops, and in different countries, limitations, scope, research trends and strategies of promotion has been discussed.

INTRODUCTION

The world is facing a serious shortage of fresh water due to dwindling resources. There is growing competition for clean water making less water available for agriculture, the largest consumer of water which accounts for more than 70% of total withdrawals for the world at large. The average irrigation efficiency of the world is fairly low. As per Gittinger (1), if this efficiency can be raised by 10%, the total conserved water will equal the water required in the world for all other uses. According to Smith (2), sustainable food production will depend on the judicious use of water resources to meet future food demands. Water productivity in terms of output per unit of food per cubic meter of water needs to be increased both in irrigated and rain-fed agriculture. In short, there is need for more crop per drop.

As urbanization, industrialization, and other uses increase, the availability of water for agriculture is likely to be reduced. By 2030, it is estimated (3) that irrigated areas will have expanded by 23%. However, decreasing water availability will allow an increase of only 12% more water for agriculture. This would necessitate an increase in water productivity by increasing water use efficiency and would require improving irrigation efficiency from the present 43% to 50% by 2030. This would in consequence require reducing irrigation water losses by introducing appropriate technologies and improved water management and cultural practices.

Water management involves various important aspects, in which there is the necessity and large scope for improvement. One of the important aspects is the field application of irrigation water. In surface application, common efficiencies range from 50–60%, whereas in drip irrigation it ranges around 90%. Although a microirrigation system appears initially costly and not necessarily applicable to similar land conditions as surface irrigation, it is definitely water saving and improves the yield and quality of crops. This article discusses microirrigation and its nature, extent, and future scope.

Historical Development

Earlier experiments on drip irrigation were done in Germany around 1860 using clay pipes having open joints, for both irrigation and drainage. In the United States, some work was done at Colorado State University in 1913. With the introduction of perforated plastic pipes in Germany, Canada, and the United States for subirrigation, this concept received quite an impetus. In these efforts, the work of Symcha Blass, an Israeli engineer is worth mentioning; he observed in 1940 that a big tree near a leaking pipe showed more vigorous growth than other trees in the area. From this observation, he arrived at the concept of irrigation, wherein water was applied in small quantity, virtually drop by drop. The earlier drip irrigation system consisted of plastic capillary tubes attached to large pipes. The experiments around 1960 at Arava in Israel showed very good results. With the development of new plastic components after World War II, great headway was made in manufacturing plastic pipes and other accessories. Drip system components were gradually properly established and sold in different countries. Per one survey, this system has been adopted in 35 countries and is installed in an area of 1,784,846 ha.

Considering the rapid growth of microirrigation, six International Congresses have been organized so far: the first in Israel in 1971, the second in California in 1974, the third in California in 1985, the fourth in Australia in 1988, the fifth in Florida in 1995, and the sixth in South Africa in 2000.

MICROIRRIGATION: DEFINITION AND TYPES

According to Nakayama and Bucks (4), trickle irrigation is the slow application of water on, above, or beneath the soil by surface trickle, subsurface trickle, bubbler, mechanical move spray, or pulse systems.

In this method, water is applied by low pressure pipes at a low rate near the root zone, in small quantities at frequent intervals. The water is applied as discrete or continuous drops, small streams or sprays, through emitters placed near the plants. It is also known as localized irrigation, drip irrigation, trickle irrigation, daily flow irrigation, and microirrigation.

Microirrigation is presently the most accepted name that includes all variations such as surface trickle, subsurface trickle, bubbler, mechanical move spray, and pulse, which are briefly described below.

Surface Trickle. In this system, the emitters and laterals are placed on the soil surface. It is the most common type and is also called drip irrigation. It is used for widely spaced as well as row crops. Discharge rates are lower than 12 L/h for single outlet, point source emitters, and less than 12 L/h/m for line source emitters.

Bubbler. In this method, water is applied to the soil surface in a small stream or fountain with a point source discharge much higher than surface or subsurface trickle but less than 225 L/h. A small basin is required for proper water distribution.

Spray. In this system, water is applied as a small spray, fog, or mist. Discharge is less than 175 L/h, and it is used to irrigate trees or other widely spaced crops.

Mechanical Move. In this system, the concept of the bubbler is extended to large spaced row crops, including travelling trickle and drag or hose reel systems.

Pulse. It has high discharge rate emitters, and thus short application times. The discharge ranges are 4–10 times larger than surface trickle emitters. This system has less clogging problems but needs inexpensive pulse emitters and automatic controllers.

Subsurface Drip Irrigation. This is another variation of drip irrigation, also covered under the term microirrigation. Its only difference from surface drip irrigation is that it is installed below the soil surface.

In this system, water is applied slowly below the surface through emitters whose discharges have the same range as surface trickle emitters. This is useful for small fruit or vegetable crops. Historically, it is the oldest modern irrigation method used in the United States. House (5) successfully used it for irrigating apples, alfalfa, and cereal crops with porous subsurface laid pipes. He, however, found it extremely costly and thus not economically feasible for field crops. The advancement in plastic materials, however has made it economically feasible for several crops. Blass (6) applied drip irrigation using laterals with buried drippers, but the practice was discontinued because of emitter clogging and root penetration. However, the practice was revived for the following reasons: (1) The same system could be used for several consecutive crops with minimum tillage. (2) The labor requirement in removing and installing the tubing with each crop was reduced. (3) There was less interference with weeding, spraying, or harvesting of the crop with buried laterals.

CROPS SUITED AND ADVANTAGES

Crop Suited

There are many crops for which drip irrigation has been used. The major fruit crops for which it has been used are citrus, deciduous, strawberry, grape, mango, pomegranate, banana, peach, pear, apricot, papaya, and coconut. Some of the important vegetables, cash crops, and plantation crops, to which it has been applied are tomato,

potato, sugarcane, cotton, olive, coconut, watermelons, cardamom, and ornamental trees.

Advantages

These are the main reasons that drip irrigation is used:

1. It provides a high degree of control for water application. Using such control, it is possible to attain much higher efficiencies of the order of 90% compared to sprinkler irrigation (60–80%) and surface irrigation (50–60%). Thus, it provides high water saving and high water use efficiency compared to other methods of water application.
2. It supplies water at a rate sufficient to satisfy evaporative demand by maintaining high matric and osmotic soil water potentials, which minimize water and osmotic stresses. In conventional systems of water application such as flood or sprinkler methods, the fluctuations in soil matric and solute potentials are comparatively large. The lesser fluctuation and thus less stresses result in better plant growth, an increase in yield, and better quality of fruits.
3. There is partial wetting of soil with drip irrigation. Because of this, less water is lost through evaporation. However, on a seasonal basis, there may not be more water saving for this reason than in conventional methods. But this helps in reducing weeds, resulting in a lower cost of weed control. It is helpful in carrying out further cultivation operations, such as spraying and harvesting, without damaging soil structure.
4. The foliage remains dry, so there is less growth and incidence of plants diseases, reducing the use of pesticides and fungicides.
5. It permits using saline water. The increase in salt concentration because of soil drying between two irrigations is less in drip than in other irrigation methods. Salts from the wetted portion are frequently leached from the active root zone. Leaf burn is avoided because there is no foliar application in this method.
6. It can be well adapted to marginal soils, such as soils with high permeability and low water holding capacity, sands, leached tropical soils, and also for steep land and slowly permeable soils.
7. It requires less agricultural chemicals such as fertilizers, herbicides, and pesticides. Because their application is according to precise plant needs, the quantity required is less, leaching losses are less, and use is more efficient. It is thus cost as well as labor saving, and causes less nitrogen pollution in groundwater.
8. It uses less energy compared to high head sprinklers.
9. It is very well adapted to greenhouses in water and nutrient application without foliage wetting, such as needed for flowers and potted plants. It is also useful for water and fertilizer application in mulched crops and with wind tunnels.

Table 1. Yield Increase, Water Saving, and Payback Period

Sl.No.	Crop	Investment, \$/(Rs.) ^a	Payback, Years	Water Saving, %	Yield Increase, %
1.	Pomegranate	625/(30,000)	1	45	45
2.	Grape	917/(44,000)	>1 year	48	23
3.	Sugarcane	989/(47,500)	1	56	33
4.	Banana	979/(47,000)	1	49	52
5.	Tomato	625/30,000	NA	39	50
6.	Chili pepper		NA	62	44
7.	Sweet lime			61	50
8.	Watermelon			40	88
9.	Okra			40	16
10.	Cabbage			60	2
11.	Cotton			53	26

^aRs. refers to Indian currency; 1\$ = Rs48 (variable with time).

Advantages of Subsurface Drip

Although another variation of microirrigation, its advantages need special mention. According to Phene et al. (7), it has the following advantages compared to surface drip: (1) There is a substantial increase in water use efficiency(WUE). (2) It enables precise fertilization with high frequency irrigation. There is negligible evaporation and almost total elimination of deep percolation and nitrate–nitrogen leaching. (3) There is long-term sustainability of the system.

WATER SAVING AND YIELD INCREASE

The most important aspects of microirrigation are water saving and increased yield compared to conventional irrigation. The experience for a few crops grown in India, as reported by Uriel (8) (slightly modified), are given in Table 1.

Table 1 shows that there is a 40 to 60% saving of water with drip irrigation over conventional irrigation. This water can be diverted to increase the area under irrigation or to other uses. Similarly, the yield increases from 23 to 88%. For vegetables and closely spaced crops, the investment is about \$979 (Rs.47000)/ha, whereas for trees, it is around \$625 (Rs.30000)/ha. For bananas, the payback is less than 1 year, whereas for fruits like pomegranate, the payback is 1 year.

COMPONENTS AND SYSTEM DESIGN

Components

Emission Devices. This is the most important component, for which sometimes the method of irrigation itself is named. This component acts as energy dissipater, reducing the inlet pressure head (0.5–1 atmosphere) to zero atmospheres at the outlet. It is generally made of polypropylene. Some types may be called orifice emitters, orifice-vortex emitters, long path emitters, compensating long path emitters, long path multiple outlet emitters, grove and flap short path emitters, grove and disc short path emitters, and twin wall emitters. The discharge rate of an emitter is characterized by its mean at normal operating pressure

and the coefficient of manufacturing variation. Its operation varies in the range of 1–2 atmosphere. In orifice type emitters, the pressure is dissipated through a small hole 0.4–0.6 mm. They are highly prone to clogging. In long path emitters, the pressure is dissipated by flow through a long narrow path. The simplest type is the microtube. The diameters of microtubes vary from 0.6–1 mm. The energy loss in long path emitters can be increased by creating a tortuous long path in the form of a spiral or labyrinth. Sometimes, the energy can be dissipated through perforations or small holes. Biwall or twin bore are such systems. In these systems, there is a small tube or a supply chamber operating at a high pressure and fitted with large, widely placed orifices. For each inner orifice, there are many outer orifices in the emission chamber. There are several in-line systems with different designs conceived by different manufacturers.

In-Line Drippers. In-line drippers are based on the long path principle. They are molded plastic accessories, in which the thread is a long narrow passage governing the slow discharge of water. Some emitters are attached to the lateral tubing and in others, the lateral tubing forms the outer wall of the tubing fixed along with the line. They may have discharges of 2–8 L/h at 1 atmosphere.

On-Line Drippers. These drippers are fixed on the lateral by punching suitable holes in the pipe. They may be classified as nonpressure compensating type or pressure compensating type.

Simple Type with Laminar Flow. In this type, the discharge is directly proportional to pressure. It has a simple thread, labyrinth, zigzag path, or other arrangement to dissipate energy.

Turbo Key Type. This is a blockage resistant and pressure compensating type, made of stabilized polymers, available in 2,4, and 8 L/h discharge.

Pressure Compensating Type. This type, provides uniform discharge between pressures of 0.3–3.5 atm. This type dripper is provided with a high quality rubber diaphragm to control pressure. It gives 2,3,4, and 8 L/h

discharge at varying pressures and is suitable for sloping topography.

Built-in Dripper Tube. In this type, the drippers are welded to the inside of the tube during extrusion of polyethylene pipes. They are provided with an independent pressure compensating discharge mechanism and wide water passage to prevent clogging.

Other Components

Lateral Pipes. These are pipes with diameters of 12–33 mm and wall thicknesses sufficient to withstand pressures of 4–6 atmospheres, to which the emitters are connected.

These pipes should be flexible, noncorrosive, resistant to solar radiation, and able to withstand temperature fluctuation. Normally, pipes are made of diameters 10, 12, 16, and 20 mm, 1–3 mm thick, and generally laid above the ground. They are commonly manufactured from low density polyethylene (LDPE) or linear low density polyethylene (LLDPE). LLDPE provides better protection from the ultraviolet rays of the sun.

Main and Submain Pipes. They are placed below the ground and supply water to the laterals. They are made from rigid PVC or high density polyethylene. The pipes may be 65 mm or more in diameter and can withstand pressures of 6 kg/cm². These pipes are laid underground for long life. For submains, rigid PVC, HDPE, or LDPE pipes are generally used; diameters are 32–75 mm and can withstand pressures of 2.5 kg/cm². They are commonly laid above ground.

Filters. This is an important component used to reduce or minimize blocking or clogging of emitters. They are of three types.

Media Filter. Required to remove organic matter such as algae. It is made up of a circular tank filled with layers of coarse sand and different sizes of gravel and valves for flushing the assembly in case of clogging. It is available in different sizes and capacities as required by the system.

Hydrocyclone Filter. Also known as a vortex sand separator. When the irrigation water has more sand, it is used as a prefilter to remove the sand before it enters the drip system. It is generally followed by a screen filter.

Screen Filter. It is fitted in series with a gravel filter to further remove impurities like fine sands, dust, etc. from the water. It consists of a single or double perforated cylinder placed in a plastic or metal container. Generally, 100–200 mesh screens are used in these filters. Screens are specified by the diameter of the pipe, recommended flow rate, total surface area of the filter, or by the cleaning method. The head loss needs to be periodically measured, and if it exceeds the permissible specified limits, it needs to be cleaned.

Fertigation and Chemigation

The process of adding the fertilizer with irrigation water is also known as fertigation. When other agrochemicals for

control of insects and pests are used along with fertilizers, instead of fertigation it may be termed chemigation. Fertilizer application through microirrigation not only helps in providing fertilizer economy, it also minimizes health concerns. However, the potential to apply other chemicals is considered a great risk. Microirrigation cannot be used for chemigation with chemicals such as fungicides, herbicides, and growth regulators, that need to be applied to plant foliage. Three methods commonly used for fertigation consist of a bypass pressure tank, a venturi, or direct injection system.

In a bypass system with the closing of the main valve, a certain fractional quantity of flow is made to pass through the fertilizer tank. This bypassed water, dissolving the fertilizer, goes to the drip system. In the venturi type, a suction head is created at the constriction which sucks the fertilizer through the system. In the direct injection type, pumps of piston or diaphragm type, operated by system pressure inject fixed quantities of fertilizers into the system.

Automation

This refers to operation of the system with minimum manual intervention and is useful in large areas. This may be done in three ways. The first method is time-based. In this method, the time of operation is estimated according to the volume of water required and the average flow rate of water. The duration of irrigation required has to be determined for each section. The duration for each valve is fed into the controller along with system start time, and the controller clock is set with the current day and time. The clock actuates the starting time of the first valve and simultaneously the pump. After the duration of the first valve is over, the first valve is closed, or the controller switches to the next valve. The second method is volume-based. In this system, the preset amount of water can be applied in the field by using automatic control metering valves. Automation using a volume-based system is of two types. In one type, the valve with pulse output provides one pulse after completing one dial of the valve. The controller accepts the pulse input and counts the volume per pulse. The volume required in a segment, can be programmed in the controller. In the second type, the valves are placed near each segment, and no controller is required. The valves are interconnected in series through a control tube. During sequential operation, only one valve remains open. The third system is real-time feedback based on the demand of the plant. The plant itself determines the degree of irrigation required. Sensors such as tensiometers, relative humidity sensors, rain sensors, and temperature sensors control irrigation scheduling. Conventional scheduling is based on refill and full point. A real-time system controls and holds soil moisture near a constant value.

System Design

Collection of General Information. This requires collection of basic data including information about water source; topography; crops to be grown; general nature of

soil and its characteristics such as texture, depth, infiltration etc.; climate; and rainfall. Such information is necessary for irrigation scheduling.

Field Layout. Depending on the type and location of the source, size, geometric layout, and topographical features, the planning is done. A contour map is helpful, but if the highest point, lowest points, ridge line, and drainage lines can be located, the layout of the system and the lengths of mains and laterals are decided.

Crop Water Requirement. The peak irrigation requirement (PIR) expressed in L/day/plant, may be estimated from the following relationship:

$$\text{PIR} = E_{\text{pan}}^* \times A \times K_p \times K_c \times K_r \times E_a + L_r - R$$

where

E_{pan}^* = (mm/day) is the average maximum pan evaporation;

A = total area allocated to each plant, m^2 ;

K_p = the pan factor ($E_{\text{to}} = K_p \cdot E_{\text{pan}}$ is reference crop evapotranspiration; K_p is less than 1 and may be taken as 0.8);

K_c = crop coefficient, selected for a given crop, growth stage and climatic conditions and has to be experimentally determined or adopted from available information;

K_r = a reduction factor, based on crop ground coverage, also known as the canopy factor varies from 0–1, and for a canopy occupying full K_r , is equal to 1;

E_a = overall application efficiency ($E_a = k_s \cdot k_u$, k_s is a coefficient, whose value is less than 1 and takes care of deep percolation and other losses, k_u takes care of application uniformity smaller than 1),

L_r = extra amount of water needed for leaching, taken as 0, where there is no salinity and no leaching problem;

R = rainfall taken as 0, where in an arid region, a crop is to be grown only with irrigation.

For many maturing crops with good water holding capacity, different factors may be taken as 1, and a rapid estimate may be made using the relationship,

$$\text{PIR} = 0.8 E_{\text{pan}}^* \times A \times E_a$$

Monthly Water Requirement. As per INCID (9), a simplified estimate may be obtained as

$$V = E_p \times K_c \times K_p \times A \times N (\text{liters})$$

where E_p is the mean monthly pan evaporation (mm/day), A is the area to be irrigated m^2 , and N is the number of days in a month.

Net volume to be applied may be written as

$$V_n = V - R_e$$

where R_e is the effective rainfall (mm).

The operating hours of a system for a month may be obtained as

$$T = \frac{V_n \times W_p}{(\text{Drippers/plant}) \times \text{plant numbers} \times \text{dripper discharge}}$$

where W_p is the wetted area.

Operating hours per application may be written as

$$T_a = T / N_m (N_m \text{ is number of application/month})$$

System Design

Laterals may be designed by considering the flow of each lateral (de) obtained by multiplying the discharge of each dripper by the number of drippers per lateral. The diameter of submains may be designed by considering the flow of each submain line (ds), obtained by multiplying by the number of laterals.

The design of a main line may be done by considering the total flow of all the submains (dm), obtained by multiplying (ds) by the number of laterals.

The design of diameters of laterals, submains, and mains may be done by considering the pressure drop due to the friction of each lateral, submain, and main using the head loss relationship of Blassius or the William and Hazen equation.

The pump horsepower may be determined from the friction head losses of mains, submains, and laterals (H_f), the static head for the pump-well system (H_s), and the operating pressure head of the emitter (H_e):

$$\text{Total head } H = H_f + H_s + H_e$$

Thus,

$$\text{HP} = \frac{H \times d_m}{75 \times M_p \times M_m}$$

where M_p and M_m are the efficiencies of the motor and the pump, and d_m is the discharge of the main in liters per second.

EXTENT OF MICROIRRIGATION

There has been a very fast rate of adopting and extending microirrigation in different countries in the last 20 years, and several surveys have been carried out to find its coverage.

From the third international survey by ICID, reported by Bucks (10), it was found that from an area of 412,760 ha in 1981, it increased to 1,082,631 ha in 1986, and 1,768,987 ha in 1991. This was a 63% increase in 5 years and a 329% increase in 10 years. The major users of microirrigation in decreasing order, according to a survey in 1991, were the United States, Spain, Australia, South Africa, and Israel. Other countries having sizable microirrigation were Thailand, Columbia, Jordan, Brazil, China, Cyprus, Portugal, Chinese Taipei, and Morocco.

The fourth survey has been in process through the endeavor of Kulkarni of ICID. Preliminary feedback has been reported from 1981–2000, 19 years, by Reinders (11)

Table 2. Microirrigation Area in a Few Countries Over 19 Years^a

Country	1981, ha	1986, ha	1991, ha	2000, ha
USA	185,000	392,000	606,000	1,050,000
India	20	0	55,000	260,000
Australia	20,050	58,738	147,011	258,000 ^b
Spain	0	112,500	160,000	230,000
South Africa	44,000	102,250	144,000	220,000
Israel	81,700	126,810	104,302	161,000
France	22,000	0	50,953	140,000
Mexico	2,000	12,684	60,000	105,000*
Egypt	0	68,450	68,450	104,000
Japan	0	1,400	57,098	100,000 ^b
Italy	10,300	21,700	78,600	80,000
Others	70,770	434,036	294,873	523,300
World	436,590	1,030,578	1,826,287	3,201,300

^a Reference 11.

^b Hectareage adjusted from 1991 figures according to the average percentage increase of other countries, as per data obtained by Dr. Kulkarni of ICID.

and is given in Table 2. From 1981–1986, a 136% increase took place and from 1986–1991, a 77% increase took place. From 1981–2000, 19 years, usage increased 633%.

Although there has been a tremendous increase in the use of microirrigation, the total area under microirrigation continues to remain only 0.8% of the total irrigated area of the world (10).

Besides the ICID surveys, according to Uriel (8), there have been several other assessments of the extent of microirrigation in different countries which are briefly given below.

According to a survey, it was estimated that in the Mediterranean coastal plains in Palestine, about 70.5% of vegetables was irrigated by drip irrigation, 28% by sprinklers, and a negligible area by traditional methods. It was found that 87% of farmers in the Jordan Rift Valley can grow two crops with modern methods in comparison with 83% in the coastal region. In Jordan, mainly in the Jordan Rift Valley, 64% of farmers use drip irrigation.

In the mountainous area around Lake Titicaca between Peru and Bolivia, a combination has been made between simple rural greenhouse and drip irrigation for growing potatoes and vegetables. Potatoes grown in greenhouses produced 350 kg/50 square meters an increase of 500% in yield in comparison to traditional methods. In Mauritius, about 33,000 small farmers cultivate about 25,000 ha of sugarcane. In several other countries like Nicaragua, southern Argentina, Thailand, Turkey, Peru, Mozambique, and China, drip irrigation is being tried for vegetables and orchard crops.

According to Smajstrla et al. (12), about 20% of the U.S. microirrigated fruit crops consist of Florida fruit crops which are primarily citrus. About 358,000 ha area in Florida is occupied by fruit crops most of which is citrus. Microirrigation provides better freeze protection and faster growth of citrus trees. Over 11% of 1.9 mha of commercial agricultural crops is microirrigated, of which 94% is in fruit crops, primarily citrus.

Among field crops, microirrigation has been extended to sugarcane and tobacco; 90% of Florida sugarcane is grown in muck soils, where subirrigation is practiced. Only 10%

of sugarcane is in sandy soil which has the potential for microirrigation.

Vegetable Crops

Vegetable crops are grown in over 140,000 ha in Florida, most of which is irrigated. About 2100 ha of strawberries grown in Florida are drip irrigated because of production benefits.

Per Chieng and Gulik (13), of 940,000 ha of land under irrigation in Canada, about 4700 ha were under microirrigation, and the trend is increasing. Most of the large scale microirrigation systems were in the province of British Columbia. In most of these areas, chemigation was being practiced, and the systems were used for tree fruit, field vegetables, greenhouse, flowers, and vines.

Drip Irrigation in some indigenous form has been practiced in India for a long time. Perforated earthenware pipes have been used for quite some time in different parts of the country such as Maharashtra and Rajasthan. Similarly, perforated bamboo pipes and pitchers have been in use. In Meghalaya, they have used bamboo drip irrigation systems for betel pepper and areca nut crops by using low flows from hill streams. Drip irrigation in modern form was introduced in the early 1970s in India. Significant development has taken place in the 1980s. Per some estimates, from a small coverage of 1500 ha in 1985, the area increased to 6000 ha in 1988, 66,000 ha in 1993, and 260,000 in 2000.

The adoption of drip irrigation has been maximum in acute water scarce areas of Maharashtra state for commercial and horticultural crops such as coconut, grapes, banana, fruit trees, sugarcane, and plantation crops. About 85% of the installed drip systems in India has taken place in Maharashtra. The states of Karnataka, Tamil Nadu, Gujarat, and Andhra Pradesh have been gradually picking up.

According to Jain (14) there were 0.38 mha of land under microirrigation, making India the second largest in the world next only to the United States which has 1.055 mha.

Extent of Application to Different Crops and Trends

Per the survey, microirrigation has been applied differently to different crops, according to Bucks (10). It has been applied most to tree crops, covering 41.5%. Among fruits, deciduous types constitute the second largest coverage having a share of 16.7%. Citrus fruits constitute the third next highest, having a coverage of 13% of all the area covered. Olives/nuts are another crop having a share of 5.1%. Among vines, it has been applied most to grapes having a share of 11.1%.

The other tree crops under major application of microirrigation consist of avocado, mango, olive, nut tree, and coconut. There has been quite an increase of microirrigation in vines and vegetables which constitute 26% of the area under microirrigation. Application of microirrigation was done for cash crops like sugarcane in Hawaii on a large scale on about 42,000 ha in 1991. Coconut occupied an area of 37,000 ha. Among vegetables, it has a maximum coverage under field vegetables.

Greenhouses are expensive, so it is understandable that the area covered under such a system would be less than that of field crops. Microirrigation has been applied to a large number of unspecified crops. The area under this category in the 1991 survey was 21.5% of the total area of 1.8 mha.

In the trend of component use, it was observed that bubblers and microsprinklers were being installed more for tree crops than individual drippers. Line source applications were being used more than individual drippers. Pressure compensating drippers were increasing in use in some countries but were generally creating problems after 2 to 5 years. Fertigation was increasing in many countries such as the United States, Spain, Australia, Israel, South Africa, and Mexico. In other countries, it was still in the initial stages. Emitter clogging was generally caused by sediments, precipitated salts, or bacterial slimes. These problems were gradually solved with bubblers and microsprinklers as well as by better fabrication. To control clogging, generally, acids and chlorine were applied. The application efficiencies ranged from 60–90%.

The costs of installation in the third survey ranged from US \$ 2000 to US \$ 7000 per ha. It was generally found that line sources (drip tapes, etc.) used in vegetable and field crops were more expensive than drippers or microsprinklers used for tree crops. Systems with average costs from \$ 2000 to \$ 4000 were economically and efficiently operated for trees, vegetables, and field crops.

Extent of Subsurface Drip Irrigation

Although economical and highly efficient, the design and laying of subsurface drip is more sophisticated and complex, and its installation fairly soil and crop specific. Thus, in spite of its being promising, its application is still in its infancy. Per the ICID third survey, it had a coverage of 54,000 ha in the United States, 2500 ha in China, 150 ha in Israel, and 2184 ha in other countries, a global coverage of 58,834 ha. According to Phene et al. (15) reporting figures from the Irrigation Journal's 1992, irrigation survey out of a total of 690.2 kha of microirrigation coverage in the United States about 76.6 kha or 11.1% was under SDI. In California SDI had grown to 61.3 kha in 1992. This constituted about 80% of all SDI in the United States. The different crops to which it has been applied were potato, tomato, strawberry, cantaloupe, lettuce, cotton, grape, hops, apples, peach, almond, walnut, turf, and ornamentals.

LIMITATIONS, RESEARCH TRENDS, SCOPE, AND PROMOTION

Limitations

Emitter Clogging. This is one of the most important problems in the use of microirrigation. This may occur due to sand, rust, microorganisms or other impurities carried by irrigation water or because of chemicals precipitating in the flow system. Root penetration also reportedly causes clogging, which occurs mostly in buried systems. Proper filtration may remove suspended particles. Suspended

organic matter and algae, etc. can be removed by media filters, whereas larger suspended materials can be removed by centrifugal separators. Appropriate chemicals can eliminate clogging due to precipitation of salts in the system.

Mechanical Damage. This may be caused by rodents, birds, animals, or careless use of implements. Chemicals are now available, which if mixed with plastics, discourage or prevent damage by rodents. With subsurface drip, there is minimum cultivation, and damage due to handling or implements can be avoided.

Salinity Hazard. With the use of brackish water for irrigation, salts tend to accumulate at the wetting front. The problems are caused in arid regions if irrigation methods other than drip are used and salts cannot be leached. The problem can be reduced by using sprinkler or flood methods of irrigation to leach the salts wherever feasible.

Lack of Microclimatic Control. This method cannot commonly provide frost protection as done by sprinklers, but a variation of microirrigation, known as microsprinklering, also provides quite effective frost protection.

Higher Costs. Compared to surface and sprinkler irrigation systems, installation costs are higher. Thus, it is feasible only for appropriate field crops and fruits, where returns are cost-effective. In some developing countries, to promote economy in using irrigation water, subsidies have been given for its installation; that has given good results in extending its adoption.

Lack of Awareness. This is a system requiring high skill in design, installation, operation, and management; it requires proper demonstration and training to extend its use to farmers in developing countries.

Research Trends

The issues and type of research in different countries vary according to need. However in the United States, which has the largest area in microirrigation, per Phene (16), research trends center around four areas: (1) determination of water requirements and irrigation scheduling, (2) nutrient management/fertigation methods and water quality, (3) system design and uniformity, and (4) expert systems and simulation models. The area of water requirement is considered important in view of the difficulty of accurate determination of water requirements and irrigation scheduling. The efforts related to this area were directed toward use of personal computers for data acquisition and control systems and electronic sensors for automation. Efforts were also directed to using plant measurements for scheduling irrigation and regional and local automatic scheduling, measuring, and calculating crop evapotranspiration as well as use of simulation models. Nutrient management was considered important in view of high nitrate-nitrogen pollution because of surface application and the scope of its reduction through fertigation. Because of

improvements in materials hardware availability, quality, and manufacturing uniformity, there is a good potential for improvement in designing microirrigation systems and developing microcomputer programs. Expert systems have the ability of facilitating the transfer of knowledge from researchers to users. The soil water-plant relationship is a complex phenomenon, in which, expert systems can play a useful role in irrigation selection, scheduling, and management

Future Scope

It may be observed that the world food requirement is likely to increase as population increases. On the other hand, water availability for food production is gradually shrinking. Water scarcity has become the single biggest threat to food security, human health, and natural ecosystems. The only way to match higher food production with declining water resources seems to be optimization of available water resources through efficient water management. Application of microirrigation has demonstrated great savings in water and fertilizer and also increased crop yield and improvement of the quality of produce. Due to this, there has been a fast growth of this technology, and the area has increased to more than double in 5 years since 1981 and to more than 4 times in the next 5 years. Such a trend shows the great possibility of further increase in the future. According to future projections, by the year 2000, the area will increase to more than eight times its coverage in 1981.

During its period of progress, its applicability has been tried for several other fruit and field crops with encouraging results. It has shown tremendous scope for flowers and vegetables and use in greenhouses and nurseries. Lots of good work has been done in landscaping, and there is a good prospect for further expansion in this area.

Success in the use of sewage as well as saline water and poor and degraded soils has further enlarged its scope. The experiments in Abu Dhabi of growing eucalyptus and casuarinas through drip irrigation using waters of salinity of 2000 mg/L and 12,000 mg/L, respectively, for stabilising sand dunes, offers great hope of greening the world's deserts.

Subsurface drip has emerged as a new promising technology, useful in further economizing water use and reducing labor, and is likely to become commercialized in the near future.

According to Uriel (8), good prospect has been observed for further expansion of microirrigation in several developing countries. In China, there is a plan for water conservation in five provinces. There is also a plan for improvement of on-farm irrigation in an area of 2,50,000 ha with sprinkler, drip, and flood irrigation. In the desert region of Nubia in Egypt, drip irrigation is expected to cover an area of 100,000 ha of vegetables and fruits. In Nicaragua, a world relief organization has established 100 model farms to introduce drip irrigation for vegetables. In Argentina, 20 model farms for drip irrigation have been developed for vegetables, flowers, and orchards. In Peru, small areas were laid for drip irrigation to grow asparagus and vine yards on 900 hectares.

In Turkey, a large scale project is in the process of improving community irrigation of which 1200 ha is for drip irrigation. In Mozambique, citrus orchards next to a dam are planned to irrigate 210 ha. In India, microirrigation has been applied in a number of states such as Maharashtra, Gujarat, Karnataka, and Tamil Nadu. But there is still a large scope for its application in other states. In the state of Andhra Pradesh, a large project of microirrigation is on the anvil.

In developing countries, however, there may be a necessity of providing subsidies in the initial stages and carrying out training and demonstrations to improve the awareness of technical staff and farmers in the interest of economizing on water use and environmental health. Besides increasing awareness, there is a great need to increase the technical competence and skill of technical people involved in installation and maintenance. Development of a strong database and providing optimum design, installation, and maintenance are great needs in developing countries and would boost microirrigation expansion a long way.

In the developed world, microirrigation offers hope to economize, reduce, and alleviate the use of excess fertilizer and the pesticide leaching hazard in surface irrigation and subirrigation and pollution of groundwater. This would help improve water quality and environmental health.

Strategies for Promotion

The future global shortage of water is very well recognized. The agricultural sector continues to be the major user of water. As the future share of water for other uses increases, the share available for the agricultural sector is likely to be further reduced. It is, therefore, essential to stress the economic and efficient use of water through microirrigation and other improved technologies of efficient water application. The developed countries have already adopted this technology to quite a large extent, though there is still scope for further adoption and extension of the more sophisticated systems such as use of automation, SDI, etc. But it is more important for the developing countries faced with increasing population and the need for more food production. Some possible approaches to meet such objectives are identified:

1. Coordinated efforts of scientists, engineers, research institutions, and industry are needed to develop appropriate technologies for field adoption of microirrigation.
2. There is a necessity to create a database on different aspects of microirrigation for specific crops/cropping pattern in different agroclimatic zones of a region. It is specially important to match the water requirement of a crop/cropping pattern to the economic viability of a system.
3. The manufacturing industry should establish a research and development base for continued improvement of the equipment to attain better performance and economy.
4. Besides working on larger systems, the industry and researchers should also develop technologies

for smaller systems suited to farmers. Small landholders have successfully used and managed drip systems for small plots.

5. It is necessary to carry out large scale demonstrations by industry, government, and nongovernment institutions on different crops in different regions to establish economic viability and promote awareness among farmers/users.
6. For the application of fertilizers and pesticides along with microirrigation, there is a necessity of generating a database on the feasibility and compatibility of such practices with a given soil water/crop condition which is duly approved by a recognized scientific body.
7. Before a microirrigation system is made available, it is necessary to ensure that the different components are manufactured according to established or adopted standards of a country, so that the materials and manufactured products give the desired performance and life for the system.
8. Microirrigation, though providing efficient and economic use of water and improving the yield of crops, is comparatively an expensive irrigation system specially suited to fruit plantation and vegetables. Besides the technical feasibility of production for a profitable return, it is also necessary to look into infrastructural aspects, such as postharvest technology, transport, and marketing as well as pricing policy. Unless the infrastructural aspect is taken care of and transport and marketing are streamlined, fruits and vegetables, being perishable products, cannot be profitably disposed of, and this expensive system cannot be sustained.
9. In many developing countries, liquid fertilizers are not indigenous and are imported; they are not affordable. Policy decisions may be necessary to enable in-country manufacture.
10. The present cost of installation may prove high for small and marginal farmers in developing countries. It needs to be examined as a policy that keeping in mind the global shortage of water, it may be worthwhile to partly subsidize the installation cost in the initial stages. Easy disbursement of subsidies that avoid delays may be desirable.

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PLANT AND MICROORGANISM SELECTION FOR PHYTOREMEDIATION OF HYDROCARBONS AND METALS

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(from *Phytoremediation: Transformation and Control of Contaminants*, Wiley 2003)

SUMMARY OF PRACTICAL IMPLICATIONS

Information on plants and other organisms that have the potential for phytoremediation of contaminants aids in the planning and establishment of removal and restoration projects for contaminated sites. Environment Canada is compiling databases on plants, bacteria, fungi, and other organisms that remediate petroleum hydrocarbons and accumulate metals and metalloids. The databases PHYTOPET (for PHYTOremediation of

PETroleum hydrocarbon contaminants) and PHYTOREM (for PHYTOREMEDIATION of Metals) are user-friendly, searchable, and summarize all the known literature. Environment Canada has also drafted a Protocol, "Environmental Technology Verification (ETV) Protocol for Metal Contaminants." The program provides suppliers of specified environmental technologies and services that meet the required criteria, with the right to use the licensed ETV verification mark.

PHYTOREMEDIATION APPLICATIONS

Phytoremediation has good potential as a flexible and cost-effective means to clean up contaminated sites. Two of the most important applications are the use of plants and other organisms to degrade petroleum hydrocarbons and accumulate metals and metalloids (hereafter the term metals covers both). Plants and other organisms extract and degrade contaminants using a variety of processes such as:

1. Phytoextraction—the capability of terrestrial plants, algae, and plant tissues to sequester toxic elements, especially metals, by uptake or biosorption. Contaminated organisms can be harvested for disposal or extraction (biomining when at commercial levels) (1–6).
2. Rhizofiltration—the use of plant roots to absorb and sometimes precipitate contaminants from polluted waters (e.g., 7).
3. Phytostabilization—the use of tolerant plants to stabilize contaminants by reducing bioavailability.
4. Phytodegradation—the use of plants and associated microorganisms to degrade organic pollutants.
5. Phytovolatilization—the use of plants to volatilize pollutants.

Information on useful phytoremediation species is limited and scattered in the literature. Environment Canada has organized information into two databases called PHYTOPET and PHYTOREM. The PHYTOPET database is a compilation and global inventory of organisms, with a focus on plants, for phytoremediation of petroleum hydrocarbons. PHYTOREM is a similar database for metals and metalloids.

Plants and other organisms have a remarkable capacity to accumulate contaminants, e.g., the common sunflower (*Helianthus annuus*) (8), Indian mustard (*Brassica juncea*) (9), alpine pennycress (*Thlaspi caerulescens*) (10,11), and brake fern (*Pteris vittata*) (12). Some of the highest accumulation of metals occurs in the New Caledonia tree (*Sebertia acuminata*) (13) and the Australian shrub (*Hybanthus floribundus*) [reported by Streit and Stumm in 1993 (14) as first reported by Harborne in 1988 (15)]. The purpose of this chapter is to describe the compilation of data for plants and other organisms that degrade hydrocarbons, or accumulate, hyperaccumulate, or tolerate metals.

PLANTS THAT TOLERATE AND SPUR DEGRADATION OF TOTAL PETROLEUM HYDROCARBONS

The database PHYTOPET specializes in plant species that spur the rhizodegradation of petroleum hydrocarbons in terrestrial and aquatic environments, including wetlands. The plants (both monocots and dicots), bacteria, protozoa, and fungi covered by the database show potential for phytoremediation of a wide range of hydrocarbons—from crude oil to polyaromatic hydrocarbons, such as chrysene and benzo(a)pyrene (Hutchinson et al. this book, Olson et al. this book). In the database, special attention has been paid to the Canadian plants from the oil-producing regions in the Western Prairie and Boreal ecozones for reclamation of oil-contaminated sites. The relationships between plants, microorganisms, and other species that can phytoremediate similar chemicals or mixtures of contaminants (metals, pesticides, and hydrocarbons) have also been considered.

The database uses many of the same fields as the PHYTOREM database for plants (exchanging information on the metal elements for information on the hydrocarbons). In addition, the database compiles information on plant-associated microorganisms. The information includes concentrations of the contaminant before and after treatment, length of treatment period, soil characteristics, age of the plant when exposed to the contaminant, and requirements for phytoremediation. Data in PHYTOPET also include tolerance to salinity and Western Canadian occurrence.

Many native grasses show promise to phytoremediate hydrocarbons [e.g., *Agropyron* spp., gramas (*Bouteloua* spp.), and buffalo grasses (*Buchloe* spp.)]. Poplar (*Populus* spp.) trees also are potentially useful. Cultivated plants with phytoremediation potential include carrot (*Daucus carota*), red fescue (*Festuca rubra*), alfalfa (*Medicago sativa*), and ryegrass (*Lolium* spp.).

PLANTS THAT ACCUMULATE METALS

Scope and Organization of the PHYTOREM Database

The global inventory of PHYTOREM covers both terrestrial and aquatic plants and other organisms (such as bryophytes, lichen, fungi, algae, and bacteria) that have potential value for phytoremediation of metals. Species included in the database have an ability or potential to tolerate, accumulate, or hyperaccumulate specific metals, or be useful as sorbents for metals.

An intensive literature search was conducted for the database, for accumulation of or tolerance to the 19 selected elements listed in Table 1. Reviews on metal accumulation provided global data (16–23). Although a large number of species are known to grow over serpentine substrates, only those specifically shown to be hyperaccumulators or to accumulate substantial levels of heavy metals are included in the database.

The database contains information on organisms that are both hyperaccumulators and accumulators (Table 2). The term hyperaccumulator was originally coined for

Table 1. Metals and Metalloids Scanned for the Development of the PHYTOREM Database

Aluminum (Al)	Cobalt (Co)	Mercury (Hg)	Radium (Ra)
Arsenic (As)	Copper (Cu)	Molybdenum (Mo)	Strontium (Sr)
Beryllium (Be)	Chromium (Cr)	Nickel (Ni)	Uranium (U)
Cadmium (Cd)	Lead (Pb)	Palladium (Pd)	Zinc (Zn)
Cesium (Cs)	Manganese (Mn)	Platinum (Pt)	

plants with greater than 1000 milligrams per kilogram (0.1 percent) nickel in dried aboveground tissues (34,35). This definition has been broadened to other elements and all parts of plants. Values for accumulation were based on either literature, or a value of 100 or 200 milligrams per kilogram of dry weight was assigned where none has been specified.

Information in the database includes geographical origin of the organisms and habitat characteristics; taxonomy; environmental effects, health effects, and uses of the species; cultivation practices; sources of material or species studied; weedy or other significant relatives; and mode of action (i.e., tolerance and accumulation). See Table 3 for details of criteria and fields. The database is worldwide in scope, but focuses on species from Canada, the U.S., and high-altitude species from warmer regions.

The database, compiled using *Microsoft Access 97*® software, allows for easy access by many database formats and programs and can be updated as needed. The database consists of three organism and two reference tables (Fig. 1a). The organism tables are (1) vascular plants; (2) algae, lichens, fungi, and mosses; and (3) bacteria. The main table of reference includes all citations that relate to concentration levels of elements accumulated or tolerated, and the second is a table of general botanical and reference works consulted. The database as designed can be searched by queries (Fig. 1b) and includes three data entry forms (botany, metal, and plant) and reporting formats for organisms and reference records.

Data sets can be quickly retrieved by applying appropriate search criteria, and viewed in tabular format. An example of a data screen for a species retrieved through the filtering process is given in Fig. 2.

The database has specific search criteria, some of which are as follows:

1. To access the elements radium and uranium, use the codes Rd and Ur respectively. Searches for accumulators or hyperaccumulators, however, use commonly used short forms for these elements. For example, the character code RaA will bring up accumulation data for radium.
2. To retrieve all records of species accumulating (A), hyperaccumulating (H), tolerating (T), or precipitating (P) an element, use the symbol for the element surrounded by asterisk in the elem_action field, e.g., “*Pb*” as the query or filter criterion. To retrieve species accumulating or hyperaccumulating a particular element, the search criterion should be a

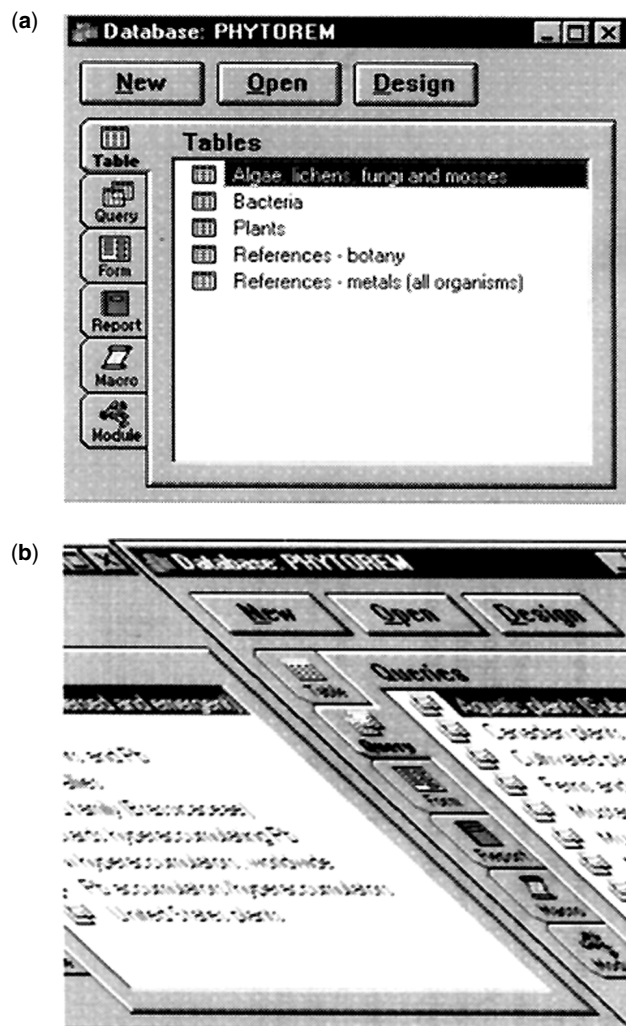


Figure 1. PHYTOREM database windows showing contents from the tables (a) and query (b) menus.

combination of the element and the desired accumulation code, e.g., “*PbH*” for lead hyperaccumulators.

3. There is no separate field to distinguish coniferous species; however, species can be retrieved by using the family Pinaceae or by retrieving all records with a growth form of TR for tree.
4. Species are reported as rhizofiltration agents for heavy metals. Studies dealing with nutrient culture experiments, although not aimed at identifying such agents, have been recorded with a Y for “yes” in the field biofiltr_use, because of the potential for such use.

Table 2. Values and Sources for Hyperaccumulation and Accumulation Criteria for Inclusion in the PHYTOREM Database

Element	Hyperaccumulation Criteria (Milligrams per Kilogram of Dry Weight)	Plants, Where Appropriate, Which Were the Basis of Criteria	References for Data	Accumulation criteria (Milligrams per Kilogram of Dry Weight)	Number of Records in the Database	Species with Highest Recorded Value {origin} [Value (Milligrams Per Kilogram of Dry Weight)]
Al	1000	Barley (<i>Hordeum vulgare</i>), horse bean (<i>Vicia faba</i>)	24	100	25	Hairy goldenrod (<i>Solidago hispida</i>) {Canada} [6820]
As	1000		25	100	4	Colonial bent (<i>Agrostis tenuis</i> = <i>Agrostis capillaris</i>) {cultivation} [2000]
Cd	100		2	10	37	Eel grass (<i>Vallisneria spiralis</i>) {India} [6242]
Co	1000		17	100	27	<i>Haumaniastrum robertii</i> {Africa} [10,200]
Cr	1000	General review	17	100	35	Alfalfa (<i>Medicago sativa</i>) {cultivation} [7700]
		<i>Dicoma niccolifera</i> ,	26,27			
		<i>Sutera fodina</i>				
		Alfalfa (<i>Medicago sativa</i>)	28			
		Smooth Water hyssop (<i>Bacopa monnieri</i>) (aquatic species)	29			
		<i>Azolla</i> spp.	30			
		Duckweed (<i>Spirodela polyrhiza</i>), kariba weed (<i>Salvinia molesta</i>)	31			
		Water lettuce (<i>Pistia stratiotes</i>)	32			
Cs	?			?	1	Sunflower (<i>Helianthus annuus</i>) {cultivation} [high absorption rate]
Cu	1000		17	100	67	Creosote bush (<i>Larrea tridentata</i>) {U.S.} [23,700 biosorption]
Hg	1000		16,17	100	35	Water lettuce (<i>Pistia stratiotes</i>) {pantropical} [1100]
Mn	10,000		33	200	28	<i>Macademia neurophylla</i> {New Caledonia} [51,800]
Mo	1500	Alpine pennycress (<i>Thlaspi caerulescens</i>)		?	1	Alpine pennycress (<i>Thlaspi caerulescens</i>) {Europe} [1500–1800]
Ni	1000		34	100	372	<i>Psychotria douarrei</i> {New Caledonia} [47,500]
			35			Shrub violet (<i>Hybanthus floribundus</i>) {Australia} [2% total ash content]
Pb	1000		17	200	79	Indian mustard (<i>Brassica juncea</i>) {cultivation} [26,200]
Sr	?			?	1	Sunflower (<i>Helianthus annuus</i>) {cultivation} [high absorption Rate]
Ur	?			?	3	Sunflower (<i>Helianthus annuus</i>) {cultivation} [> 15,000]
Zn	10,000		17	500	48	Alpine pennycress (<i>Thlaspi caerulescens</i>) {Europe} [52,000]

? indicates that criteria have not been defined for hyperaccumulation or accumulation of these elements.

Note: No reports were found for accumulation of beryllium, palladium, platinum, and radium.

Table 3. Fields, Description, and Access for PHYTOREM Database

Name	Description [and explanatory comments]	Plants	Bacteria	Algae	Access
Entry_seq	Automatic counter field to record sequence of data entry [allows multiple records taken from a single summary document to be grouped together for purposes of review and data management]	✓	✓	✓	
Type_org	Vascular plant = VP, bacteria = BA, algae = AG, lichens = LI, fungi = FU, and bryophytes = BR [field allows tables of different organisms to be combined and provides the ability to distinguish records by type of organism]	✓	✓	✓	
Growth_form	For plants only: fern = FE, graminoid = GR, herb = HE, shrub = SH, succulent = SU, vine = V, and tree = T	✓			
Sci_name	Complete scientific name with authorities	✓	✓	✓	✓
Synonym	Common synonym	✓		✓	
Com_name	Common English name(s)	✓	✓	✓	
Cv_strain	Cultivar or strain name, or code; also transgenic variants	✓	✓	✓	
Family	Taxonomic category: family	✓		✓	✓
Order	Taxonomic category: order	✓		✓	✓
Subclass	Taxonomic category: subclass	✓		✓	✓
Class	Taxonomic category: class	✓		✓	✓
Duration	For plants only: A(nnual), B(iennial), A/B, P(erennial), A/P, and B/P	✓			
Origin	Country or region of origin of plants on which report(s) is based [Canadian and U.S. species or those of other continents or countries can be sorted]	✓	✓	✓	✓
World_range	World range of the species	✓			
Primary_habitat	T = terrestrial, A = aquatic, T/A = terrestrial and aquatic [provides the ability to distinguish between terrestrial, aquatic, marsh, and wetland species (T/A)]	✓		✓	✓
Hab_descr	Habitat description [allows for sorting to distinguish plants from serpentine soils and to distinguish cultivated species (crop plants)]	✓			
Namer_occ	North American occurrence: N(ative); E(xotic); X (not present); ? (status unknown) [allows North American species to be categorized]	✓			
Namer_spp	Indication of whether other species in the genus are present in North America	✓			
Sig_relatives	Other species of significance in the genus	✓			
Cult_wild	For plants: crop plant = C, horticultural species = H, wild = W.	✓		✓	✓
Cult_info	For algae and other: cultured = C or wild = W Propagation and/or test studies, or experimental conditions	✓		✓	
Cult_source	Source of cultivated material as indicated in publication	✓	✓	✓	
Impact_attributes	UN = unknown, WD = weed, NX = noxious properties, HY = hybridizes, and PH = disease and insect pest host	✓			
Impact_description	Documentation of potential impact [information on invasive potential of an exotic weed]	✓			
Uses	General uses of plant and citations [medicinal or edible]	✓			
Bio_notes	For plants: notes on such topics as pollination and dispersal mechanism. For bacteria: notes on important and interesting aspects of biology.	✓	✓		
Gen_notes	For plants and others: toxicity of metal to the plant or other organisms and other pertinent information and citations. For bacteria: general notes on procedures of the study or results and citations.	✓	✓	✓	
Elem_conc_ref	Citation and concentration of elements with organ of storage	✓	✓	✓	✓
Elem_form_ref	Element form if specifically indicated [e.g., Cu(II)]	✓	✓	✓	
Elem_chel_ref	Element, chelate used and citation	✓	✓	✓	

Table 3. (Continued)

Name	Description [and explanatory comments]	Plants	Bacteria	Algae	Access
Elem_action	Hyperaccumulate = H, accumulate = A, tolerate = T, rhizosphere concentration = R, and precipitate = P [e.g., PbH is lead hyperaccumulation]	✓	✓	✓	✓
Storage_sites	For plants: sites where element concentrations were measured [e.g., root, shoot, or leaf]. For bacteria: sites where element measured [e.g., cell wall].	✓	✓		
Biofiltr_use	Use for biofiltration: yes (Y) or no (N) [used to distinguish species useful for such a function as indicated in the publication]	✓	✓	✓	
Biofiltr_ref	References to the biofiltration publications [generally the same as the primary reference]	✓	✓	✓	
Tolerance_info	Information on conditions under which tolerance occurs	✓		✓	
Summary_ref	Reference to summary papers where several to many species are listed	✓	✓	✓	✓
Primary_ref	Primary literature reference [obtained from summary reference if summary document was used]	✓	✓	✓	✓

Figure 2. Example of a plant report form from a search of the PHYTOREM database.

Summary of the Database Contents

The database provides an extensive, though preliminary, collection of records on phytoremediation potential. At present, the database contains records of 775 species spread over 76 families, 39 orders, 9 subclasses, and 2 classes (dicots and monocots). Data come from 39 countries of origin (based on names used in the publications).

The greatest proportion (465) of species recorded in the plant table accumulate, hyperaccumulate, or tolerate a single metal of the 19 elements surveyed. Some species take up two elements (66) or three (25), with a balance of 15 species capable of accumulating four or more elements. Table 4 summarizes the species in the latter category with elements and degree of accumulation.

The highest number of records (260) was found for nickel followed by lead and copper (Fig. 3). No records were found of species accumulating beryllium, palladium, platinum, or radium. Species with the highest concentration of the elements recorded in the database (listed in the right-hand column of Table 2) provide some

interesting possibilities for phytoremediation, including several commonly cultivated species such as sunflower (*Helianthus annuus*) and alfalfa (*Medicago sativa*).

Data Update Recommendations

Updates that could be considered include (1) information from a more extensive literature search, (2) fields for data on storage sites (e.g., roots), (3) mechanisms of tolerance or accumulation of metals by plants, (4) species that remediate radionuclides, (5) plants that remediate organic and other inorganic compounds and stabilize contaminated sites, and (6) expansion of data for algae, in particular those with a high absorptive capacity. In addition, the database could also provide pictures of plants and sites.

INORGANIC PHYTOREMEDIATION VERIFICATION PROTOCOL

Environment Canada is formulating a protocol to establish the limits of applicability for the phytoremediation of metals and metalloids in contaminated soil and water that will aid economic adoption and sustainable use. The pollutants (no organic contaminants to this point) are arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni), selenium (Se), silver (Ag), and zinc (Zn). Use of the verification protocol will allow the use of a Canadian Environmental Technology Verification symbol in remediation products. The applicant will be required to establish that a particular application has been tested (and under what conditions) (e.g., proof of principle at the bench, greenhouse, plot, or full-scale testing). Only applications that involve phytoextraction, rhizofiltration, phytovolatilization, phytostabilization, and combinations of these physiological processes will be certified initially. All phytoremediation applications (or technologies) will be evaluated based on the following criteria and information:

1. Total amount, concentration, and speciation of elements in contaminated soil, rhizosphere soil water,

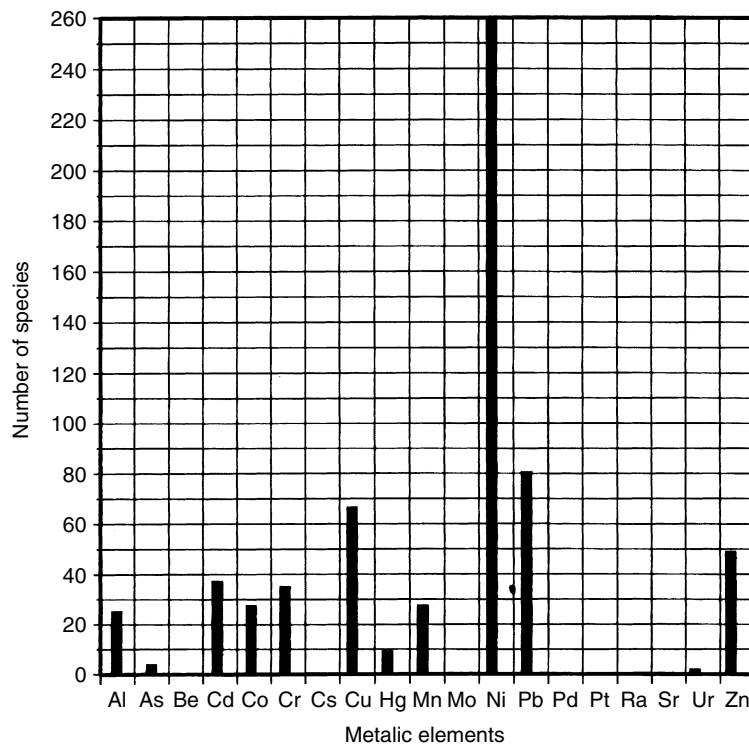


Figure 3. Number of species records for the 19 elements scanned.

Table 4. Plants Capable of Accumulating Four or More Metals

Scientific Name	Common Name	Origin and Characteristics	Elements and Degree of Accumulation ^a
<i>Azolla filiculoides</i>	Water fern	Africa, floating	CuA, NiA, PbA, and MnA
<i>Bacopa monnieri</i>	Water hyssop	India, emergent species	HgA, CuH, CrH, PbA, and CdH
<i>Eichhornia crassipes</i>	Water hyacinth	Pantropical/subtropical, troublesome weed	CdH, CrA, ZnA, HgH, PbH, and CuA
<i>Hydrilla verticillata</i>	Hydrilla	Southern Asia but introduced and spreading as a troublesome weed in the warmer states of the USA	CdH, CrA, HgH, and PbH
<i>Lemna minor</i>	Duckweed	Native to North America and widespread	PbH, CdH, CuH, and ZnA
<i>Pistia stratiotes</i>	Water lettuce	Pantropical and native to southern U.S. but an aquatic weed	CuT, CdT, HgH, and CrH
<i>Salvinia molesta</i>	Water fern	India	CrH, NiH, PbH, and ZnA
<i>Spirodela polyrhiza</i>	Giant duckweed	Native to North America	CdH, NiH, CrH, PbH, and ZnA
<i>Vallisneria americana</i>	Tape grass	Native to Europe and North Africa but widely cultivated in the aquarium trade	CuH, CdH, CrA, and PbH
<i>Brassica juncea</i>	Indian mustard	Cultivated	PbH, PbP, ZnH, NiH, CuH, CrA, CdA, and UrA
<i>Helianthus annuus</i>	Sunflower	Cultivated	PbH, UrH, SrH, CsH, CrA, CdA, CuA, MnA, NiA, and ZnA
<i>Agrostis castellana</i>	Bent grass	Portugal	AsH, PbA, ZnA, MnA, and AlA
<i>Thlaspi caerulescens</i>	Alpine pennycress	Europe	ZnH, CdH, CoH, CuH, NiH, PbH, and CrA
<i>Athyrium yokoscense</i>	Fern	Japan	CuH, CdA, ZnH, and PbH

^aH: hyperaccumulator; A: accumulator; P: precipitator; and T: tolerant.

water below the root zone, groundwater, wastewater, roots, shoots, and air (where phytovolatilization is a factor) during and after treatment

2. Amount and timing of supplements added to treated soils and waters as related to mobilization or immobilization of contaminants, including co-contaminants, and expected effects
3. Seepage or infiltration volumes from treatment wetlands or soil plots and concentrations of contaminants and by-products for the range of expected moisture conditions
4. Range of meteorological conditions for effective phytoremediation application including, but not limited to, seasonal precipitation, air temperatures, humidity, wind speed and direction, and solar radiation
5. Time required to achieve cleanup or risk-reduction standards for various levels of initial contamination in soil or water, plants selected, soil or sediment conditions, and weather
6. Soil, sediment, or water quality during and after the treatment, including texture, mineralogy, moisture content, cation-exchange capacity for soils, organic carbon and nutrient content, pH, and redox potential for all media
7. List of plants and description of communities and any rooting depth, growing habits, and foliage requirements for effective cleanup
8. Unusual ecological conditions to avoid including endangered and threatened plants and wildlife, weeds, invasive species, bioaccumulation (in the absence of netting, fencing, motion detectors, or alarms), and wetland changes (e.g., drying up of seepage wetlands)
9. Risk-reduction or cleanup levels achieved in soil, groundwater, and air for different land uses
10. Residual management options (e.g., land filling in hazardous waste landfills, composting, and pyrolysis) and any risks during handling, transport, and disposal of residuals
11. Closure plans and options for vegetative cover

These criteria will be focused on specific issues for terrestrial and aquatic remediation. Criteria and information requirements will be weighted, depending on the application. This protocol is different from that used in the U.S. (36,37); however, the differences are expected to diminish over time with continued collaborative projects between Environment Canada and the U.S. Environmental Protection Agency.

CURRENT AND FUTURE EXPECTATIONS

Plant-based remediation and restoration applications are expected to receive increasing attention in Canada and elsewhere as options for removal of the more recalcitrant inorganic and organic constituents at contaminated sites. These Environment Canada databases and protocol could provide valuable assistance in the pre-selection of

organisms, help clarify the regulatory responsibilities of proponents in securing necessary approvals prior to the use of specific organisms, and provide valuable insight into genetic materials to adapt or modify representative Canadian plant species to the climatic circumstances unique to specific sites. Future initiatives include (1) database expansion to include one for radioactive materials (PHYTORAD); (2) a Canadian network to identify, store, protect, propagate, and possibly distribute germplasm of candidate plant cultivars; (3) protocols that outline responsible provisions; (4) methods to recover inorganic contaminants from contaminated biomass; (5) a complete overview of the regulatory issues inherent in the use of plants as remediation and restoration agents; (6) definition of bioavailability and the role of plants in natural attenuation; and (7) establishment of mechanisms to limit phytoremediation site access by herbivores and omnivores.

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NITRATE POLLUTION PREVENTION

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Groundwater and springwater are sources used for water supply in many countries. Pollution of surface and groundwater is caused mostly by the chemicals used in industry, urban wastes, and agriculture. The pollution of **ground water** by **nitrates**, as well as pollution of **crops** that we consume, is one of the most serious environmental concerns in most countries. A primary characteristic of most agriculturally related groundwater pollution is that it is often widely distributed (nonpoint source), whereas localized animal farms, agrochemical and crop waste disposal and storage, industry, and urban wastes may lead to point pollution.

Using high amounts of nitrogen in fertilizers leads to an increase of nitrate in plants, where a part of the nitrate, not absorbed by plants, is washed away by rain and pollutes groundwater. Drinking water and eating vegetables with high amounts of nitrate and nitrite are toxic specially for children and babies because it may cause methemoglobinemia.

Acceptable daily intake (ADI) of nitrates is 0–5 mg/kg of body weight and of nitrites is 0–0.2 mg/kg of body weight. Children younger than 6 months are not allowed to consume food that contains nitrites. (Standards

are set by common Committee FAO/SZO for nutrient additives—JECFA.) The limit values, that are supposed to ensure safe amounts of toxic elements for the majority of people have only relative value. The well-known statement by Paracelsus (1493–1541) is, “dosis sola facit venenum” (dose determines if a substance is toxic or not). All these concepts fail for substances with irreversible, especially carcinogenic or mutagenic effects. For carcinogenic and mutagenic substances, it is evident from the viewpoint of toxicology that prohibiting consumption is needed; any amount of such substances consumed are supposed to be a risk for human health.

According to the EU Nitrate Directive, the maximum standard for nitrate in drinking water (according to the value adopted by WHO) is 50 mg NO₃/L of water.

Bottled water is not supposed to contain any NO₂, whereas regular drinking water can contain up to 0.005 mg/L of NO₂-N under normal conditions and not more than 0.05 mg/L of NO₂-N in extraordinary circumstances.

When considering the possibility of pollution of ground-water by nitrates due to surpluses of nitrogen in the soil, it is essential to touch on the reasons for nitrate surpluses. The quantity of nitrate in vegetables (freshly edible crops) depends on the relative speed of two physiological processes: absorption of nitrates by the roots of plants and protein synthesis with consumption of nitrate and its reduction to nitrite and to ammonium. If the speed of absorption of nitrate is higher than the rate of protein synthesis, nitrate automatically accumulates in plants. On the other hand, if nitrate is transformed into protein in step with its absorption, nitrate is present only in small amounts. A disproportion between nitrogen absorption and protein synthesis may be caused for two reasons which sometimes are synergistic: (1) excessive quantities of nitrogen in the soil and (2) slowing down of protein synthesis.

The nitrate level in plants depends on the type of soil, climate conditions (water in the soil), soil cultivation practices, age of plants, and type of vegetable (cultivar and variety). The nitrate level is always higher in young plants. An excess of nitrate in the soil results above all from chemical nitrogenous fertilizers particularly in the form of nitrates, which are easily available to vegetables. Several factors affect the intensity of protein synthesis; the most important are the following: light, the state of the foliage, the lack of minerals, particularly of trace elements, and the use of certain pesticides. The feebler the light, the richer plants are in nitrate (crops under glass, particularly out of season, shady locations or overcrowding, short days, and weak sunshine). If leaves are insufficiently developed or damaged by parasitic attack, the nitrate content in the roots of plants is raised. Lack of minerals slows down protein synthesis which affects the accumulation of nitrates. All factors that obstruct normal biochemical processes in the soil or in plants can slow down protein synthesis and therefore accelerate the accumulation of nitrates for example, lack of water in the soil, excessive water in the soil due to inadequate drainage, inadequately cultivated soil, and use of herbicides.

It is not nitrate that is toxic but the compounds derived from it, nitrites and nitrosamines. There are two reasons that nitrate derived compounds are toxic:

1. Nitrates oxidize the ferrous ion of hemoglobin into ferric, methemoglobin is formed, and transport of oxygen is hindered (methemoglobinemia).
2. Nitrosamines are carcinogenic.

Nitrites and nitrosamines are formed during the microbiological process that takes place immediately after harvest whenever vegetables—medium rich in nitrate—are stored under anaerobic conditions at ambient temperature (sawerkroust, etc.). The very same processes take place in cooked vegetable stored at ambient temperature.

Many years of experiments have been conducted at the Agrohydrologic Research Station in Ljubljana, Slovenia (B. Maticic), on the impact of different water application levels in the soil and different levels of mineral nitrogen in the soil on the content of nitrate, nitrite, and ammonium on crop yield for many different vegetables. To prove the importance of sustainable nitrogen fertilization of crops and optimal treatment of crops with water (irrigation), we present some results of the interaction between nitrogen application and water application for three different vegetable crops: cabbage, red beet, and celery. The results of “water–nitrogen fertilizer–yield” production functions as well as “water–nitrogen fertilizer–nitrate” functions for these three crops are presented in Figs. 1–3.

The increase of water application levels caused a decrease in NO_3 content in vegetables until the water application reached a certain optimal level; a further increase of water application also increased NO_3 content. The influence of water application (irrigation) on NO_3 content is greater than the influence of fertilization with

nitrogen. Extreme amounts of water (minimal, when plants are suffering from drought and maximal, when plants have too much water for normal growth) increase the level of NO_3 in vegetables.

It is obvious that fertilization with nitrogen increases the nitrate level in vegetables. For red beet, the maximal increase in nitrate level, caused by fertilization with nitrogen, was 450%; for cabbage only 60%. The increase

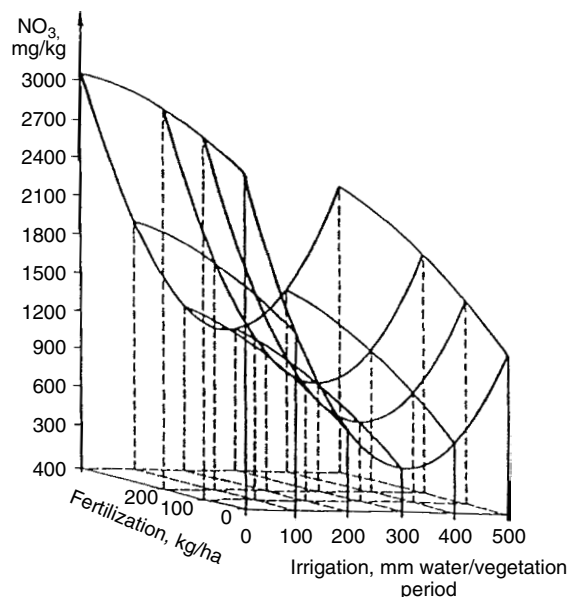


Figure 2. Nitrate content in red beet due to different water and different nitrogen treatments.

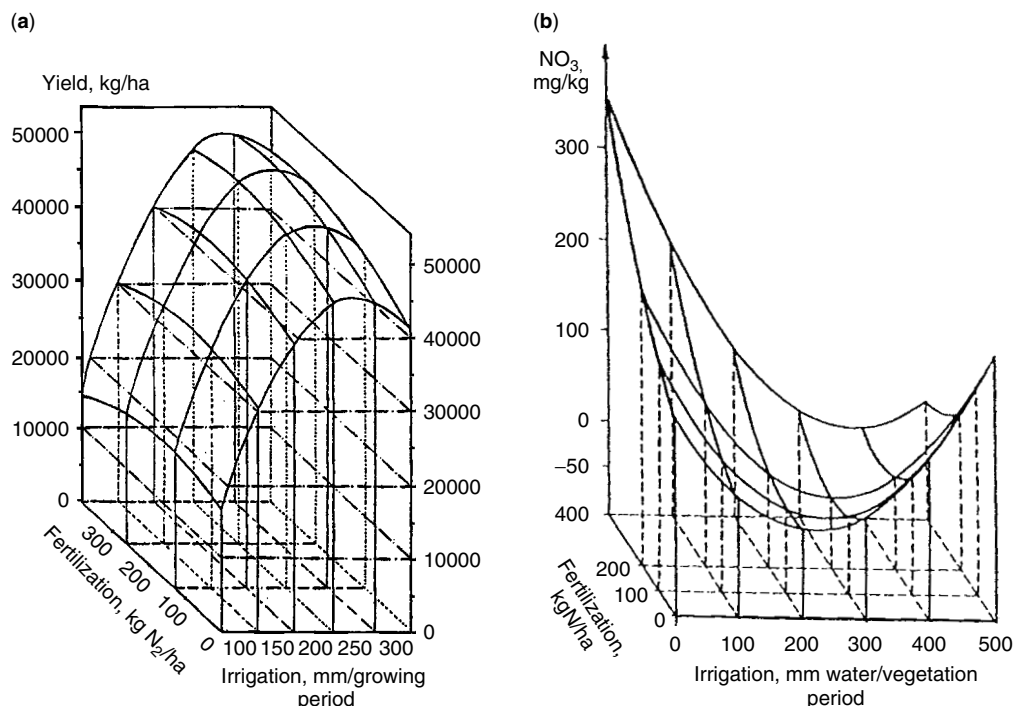


Figure 1. The impact of interaction between water application (irrigation) and nitrogen fertilization on crop yield (cabbage) and content of nitrate in the crop: (a) yield of cabbage due to different water and nitrogen treatments; (b) nitrate in cabbage due to different water and nitrogen treatments.

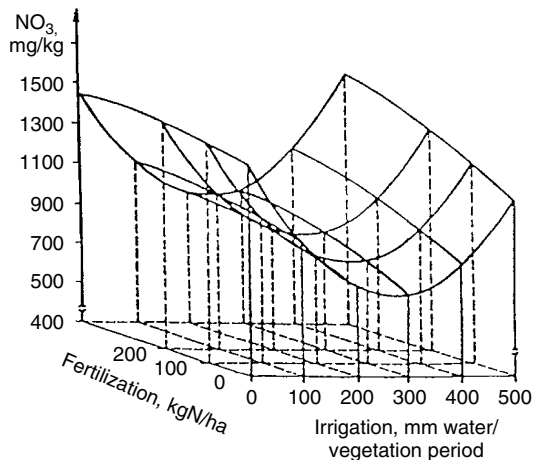


Figure 3. Nitrate content in celery due to different water and different nitrogen treatments.

in nitrogen doses from 0 to 400 kg N/ha caused an NO_3 content more than fifteen times higher in celery leaves and more than ten times higher in celery bulbs.

A significant reduction of nitrate and nitrite levels in vegetables was observed with optimal water supply and with small to medium doses of nitrogen fertilizers. According to the results, the recommended allowable water depletion from the soil for vegetable growers would be to maintain soil water conditions within the range of 20–30% of available water in the soil, and nitrogen fertilizer should be applied in several small doses.

To identify nitrogen surpluses (that may influence the pollution of groundwater and surface waters) due to agricultural technology used in different countries, it is advisable to demonstrate a **“mineral (nitrogen) balance”** on regional and farm levels using a **normative approach**. This method takes into account nitrogen input from mineral fertilizers, animal wastes, and the deposition from the atmosphere reduced by nitrogen uptake by harvested crops and ammonia losses to the atmosphere (30%).

It is also recommended to do thorough surface and groundwater quality **monitoring** according to the methodology recommended by international organizations with as many sampling points as possible for basic physical, chemical, and bacteriological analyses.

A mineral (nitrogen) balance provides information closer to the nitrate problem. Ammonia losses during storage do not directly affect groundwater. However, deposition from the atmosphere must be taken into account because it has an impact on nitrate concentration in groundwater. The average nitrogen deposition from the atmosphere is 20 kg nitrogen/ha (or 15.5 kg N/ha according to RIVM, The Netherlands). The nitrogen balance at the farm level is based on the so-called surface balance approach. The input factors include minerals provided by purchase of fertilizers and manure, minerals from production of animal manure, and deposition from the atmosphere. The output factor includes nitrogen from harvested crops. The assessments are based on elaborations by the Federal Agricultural Research Center, Braunschweig, Germany.

To reduce mineral (nitrogen) surpluses in agriculture and to meet the standards of nitrate and nitrite in drinking water and crops, it is necessary to control water and food quality in connection with nitrates and nitrites and to exact certain regulations regarding change in agriculture, especially animal excrement management. These regulations should give different norms for sustainable behavior in agricultural technology according to “Nitrate Directives” and “Code of Good Agricultural Practice.” The most important norms should be as follows:

1. The highest quantity of manure allowed on agricultural land as well as limitations for the use of the manure in specific soil conditions:
 - The maximal allowed intensity of raising animals should be 3 LU/ha for cattle or 2 LU/ha for pigs and poultry.
 - Application of organic manure should not be allowed during winter on frozen soil.
 - Application of organic fertilizers should not be allowed on soil saturated with water.
 - Application of organic fertilizers should not be allowed in temporarily flooded areas.
 - Application of organic fertilizers should not be allowed near streams (10 m away from the stream) and in depressions where there is no runoff of water.
 - Applying liquid manure on bare soil should not be allowed in the period from Nov. 15 till Feb. 15.
 - Application of organic fertilizers on water aquifer protected areas should be very restricted and done only in agreement with local authorities and regulations valid for those areas.
 - In the vicinity of springwater and in underground water pumping areas, wastewater should not be drained to springwater or underground water in any case.
2. The highest quantity of N, P_2O_5 , and K_2O allowed per hectare should be 170 kg N, 120 kg P_2O_5 and 300 kg K_2O .
3. Animal wastes should be stored in suitably arranged dung yards and cesspools so that there is no danger of leaking through and pouring over underground water.
4. The adjustment of farms to these regulations should be done as follows:
 - the adjustment of the number of animals (LU) according to the area of land available on the farm,
 - possible rent of additional land according to the contract, the construction of necessary dung yards and cesspools for hard and liquid manure, according to the restrictions.

Implementation of these regulations should be controlled by strict agricultural inspection in each country.

In countries where groundwater is relatively deep and drainage systems are supposed to drain “perched” water tables and to consider the hydraulics of water

flow in the soil profile, drainage can prevent groundwater from pollution by minerals and pesticides that contaminate the upper layer of the soil. Polluted water flows to open canals (collectors) and further to rivers. In order not to pollute water in the rivers, it is an idea (B.Maticic) to install phytoremediation lagoons at the end of open canals (collectors) before water is discharged into surface water streams (rivers). This action would undoubtedly protect the quality of groundwater and water in rivers, and therefore act as an environmental measure.

When the world is going through many changes regarding exploitation of its natural resources and environmental protection, a new idea for efficient and sustainable draining of soil is of great importance. It provides the opportunity to obtain a common goal a "sustainable development."

For many countries, where agriculture is a very important part of the economy, efficient and sustainable water management of agricultural land is very important. In the process of the transition of our society, there is strong emphasis on sustainable development, and it is believed that the field of agriculture is especially sensitive in this regard. That is why a big effort to overcome the mistakes of past development is being put forward to establish a whole new approach to agricultural planning and production with emphasis on the rational use of natural resources and modernized use of drainage and irrigation.

A determination exists to change old attitudes toward agricultural areas as underdeveloped areas, used only for mass food production, regardless of impacts on the environment. We believe that by careful planning and protection of the fertile soil in agricultural areas and forests as well as water resources, it is possible to achieve living conditions which would be attractive to people to remain on their land and continue farming traditions, as well as to the others to enjoy the environment of the countryside.

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NITRIFICATION

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Nitrification is the biologically facilitated oxidation of ammonium (NH_4^+) to nitrate (NO_3^-). The process of nitrification is performed by ammonium and nitrite (NO_2^-) oxidizing bacteria found in soil and waters (Table 1). In soil and water, ammonium is supplied through decomposition of organic matter by microbes and/or anthropogenic or natural additions of inorganic nitrogen sources containing ammonia (NH_3), urea, or ammonium. Nitrification consists of two independent microbially facilitated reactions. In the first reaction, *ammonium* oxidizers oxidize ammonium to nitrite, and in the second reaction, *nitrite* oxidizers oxidize nitrite to nitrate.

The purpose of nitrification (microbial facilitated conversion of ammonium to nitrate) is to provide nitrifying bacteria the energy they require for metabolism. The nitrification reactions are represented in Fig. 1, and the bacteria involved are listed in Table 1. There are two different microbially mediated nitrification reactions.

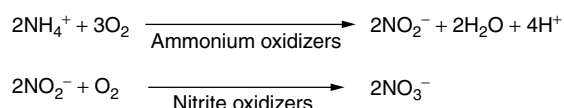


Figure 1. Nitrification reactions.

Table 1. Common Species of Nitrifiers and their Habitats^a

Bacterial Species	Habitat
Ammonium oxidizers	
<i>Nitrosomonas europaea</i>	Soil, water, sewage
<i>Nitrospira briensis</i>	Soil
<i>Nitrosococcus oceanus</i>	Marine
<i>Nitrosococcus mobilis</i>	Marine
<i>Nitrosococcus nitrosus</i>	Marine
<i>Nitrosolobus multififormis</i>	Soil
<i>Nitrosovibrio tenuis</i>	Soil
Nitrite oxidizers	
<i>Nitrococcus mobilis</i>	Marine
<i>Nitrobacter gracilis</i>	Marine
<i>Nitrobacter winogradski</i>	Soil
<i>Nitrobacter agilis</i>	Soil, water

^aRef. 1.

Chemoautotrophic nitrification is the use of carbon dioxide and carbonates as a carbon source for cell construction and oxidation of inorganic molecules such as NO_3^- or NO_2^- as an energy source. Chemoautotrophic nitrification is carried out only by a select group of bacteria. Heterotrophic nitrification is the use of organic compounds for both a carbon and energy source. Heterotrophic nitrification is executed by a wide variety of heterotrophic bacteria and fungi (2). Chemoautotrophic nitrification is believed to be up to 1000 times faster and is better understood than heterotrophic nitrification. Chemoautotrophic nitrification dominates in neutral to alkaline agricultural soils, whereas heterotrophic nitrification dominates in acid forest soils (2).

RELATIONSHIP BETWEEN SOILS AND WATERS

Nitrification is a key component of the nitrogen cycle (Fig. 2). The nitrogen cycle represents the fundamental transformations of nitrogen in the environment. In agricultural soil systems, we strive to manage this cycle better to increase crop production and minimize the negative impacts of nitrogen on the environment. Implementing management practices that will help manage nitrogen in agricultural systems to protect surface and groundwaters is an example. Nitrate, the end product of the nitrification process, is highly soluble in water and is not readily adsorbed to soil particles; therefore, if nitrate is not used by plants or microbes as a nutrient, it can move vertically through soils to groundwater and in runoff to surface waters resulting in problems from nutrient enrichment. Because nitrification is a microbially mediated process, many management practices aimed at reducing nitrate leaching focus on the nitrification process.

FACTORS INFLUENCING NITRIFICATION

Assuming an adequate supply of ammonium and a sufficient population of nitrifying bacteria, there are several factors that affect nitrification in natural systems.

These factors include pH, moisture, temperature, and oxygen concentration.

pH

Soil and water pH influence the availability of many essential nutrients such as calcium and phosphorus (3). Microbes require these nutrients for growth and development; therefore, pH ranges that promote adequate availability of these nutrients are required for nitrification. Nitrification in soils is optimized at pH 8.5 but can proceed in a pH range of 4.5 to 10 (3). Figure 3 illustrates the relationship between soil pH and nitrification. At low pH levels, many essential nutrients are less available for use by plants and microbes.

Moisture

Nitrifying bacteria require water to live and function. In soils, the optimum water content for nitrification is at field capacity (water remaining in soil after gravity has removed excess water after a period of saturation). Research shows

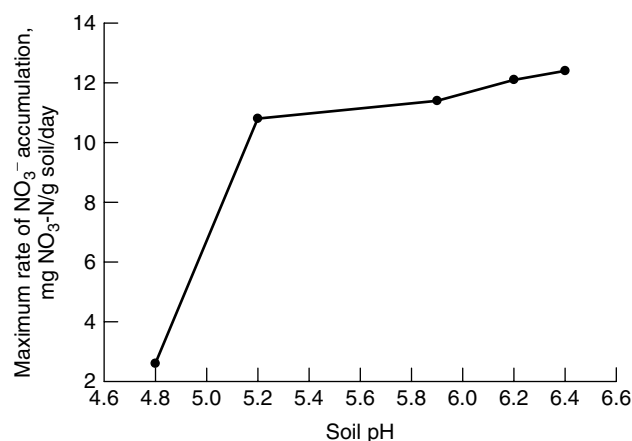


Figure 3. Effect of soil pH prior to incubation on nitrate accumulation in soil treated with 169 kg N as NH_4NO_3 /ha and incubated at 23 °C for 30 days (4).

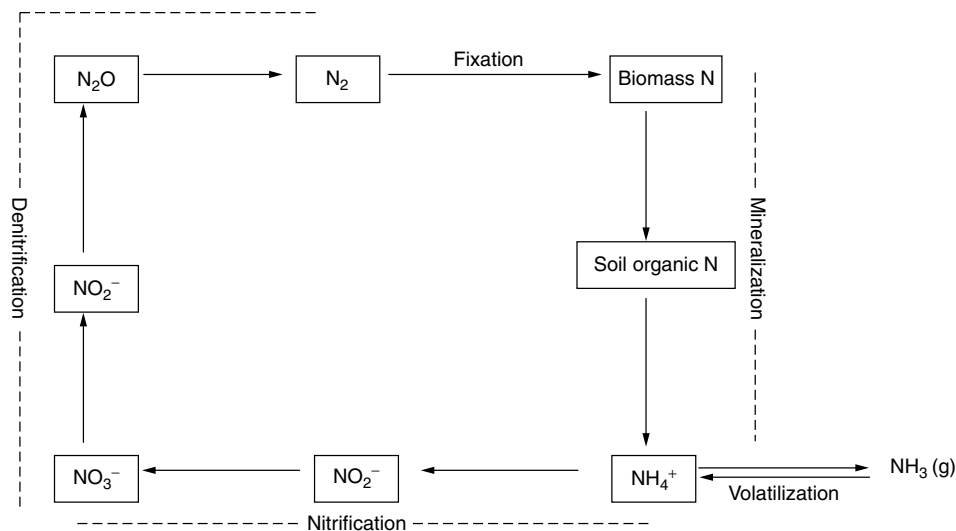


Figure 2. Nitrogen cycle.

that, as water content increases in soil from dry to an optimum content, nitrification rates increase (Fig. 4). However, waterlogged conditions drastically decrease oxygen content and thus decrease the rate of nitrification. The water content at field capacity varies, depending on the soil physical properties such as texture and organic matter content.

Temperature

Generally, as temperature increases to a point so does the rate of nitrification (Fig. 5). Studies have shown that ammonium is converted to nitrite at temperatures from 0°C to 65°C and nitrite is converted to nitrate from 0°C to 40°C. Rates of nitrification dramatically decrease above and below these ranges. Optimum temperatures for nitrification are between 30°C and 35°C.

Oxygen Concentration

Figure 1 illustrates the need for oxygen in the nitrification process. Diffusion rates and concentrations of oxygen

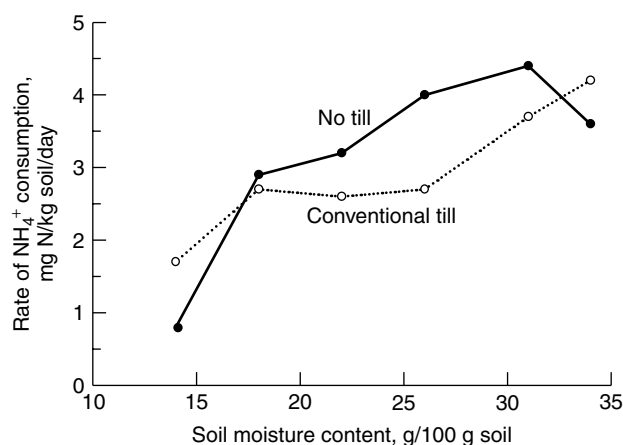


Figure 4. Nitrification rate of two tillage systems of a Maury silt loam soil column related to soil moisture content (5).

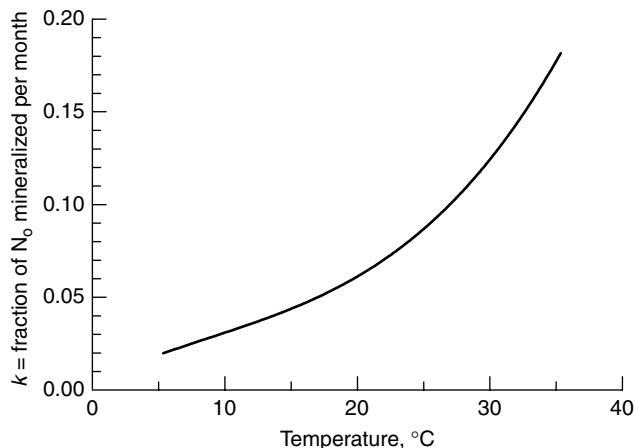


Figure 5. Fraction of N mineralized per month, k , in relation to temperature (k was estimated graphically for observed average monthly air temperatures) (6).

in soils and water will directly influence the rates of nitrification.

MANAGEMENT OF NITRIFICATION IN SOIL TO REDUCE NITRATE LOSSES TO SURFACE AND GROUNDWATER

Reducing nitrate losses in agricultural soils that receive large nitrogen (ammonia and ammonium) input from fertilizers is important to maintain high quality water resources. Chemicals called nitrification inhibitors have been developed that inhibit the nitrogen oxidative processes of nitrifying bacteria when added to agricultural soils concurrently with ammonium based fertilizers. In other words, nitrogen in the form of ammonium persists in the soil for longer periods of time that allow for increased plant use of the ammonium. Ammonium is adsorbed to soil particles; therefore, it does not readily move with water like nitrate. Nitrification inhibitors have a finite effective functionality in soil because soil microbes degrade them over time. Studies have shown that nitrification suppression by the inhibitor dicyandiamide can last 3 months, depending on the amount added with fertilizer (3).

Management practices that use knowledge of the factors that influence nitrifiers are common. For example, most farmers apply their ammonium based nitrogen fertilizers late in the fall when soil temperatures have declined to a point where nitrification is inhibited. If the farmer applies the fertilizer when soil temperatures are sufficiently high, then the ammonium is nitrified to nitrate and rainfall and/or snowmelt can leach the nitrate out of the root zone to the groundwater before the next crop can use the nitrogen.

Nitrification is a natural biological process involved in the transformation and fate of nitrogen in the environment. However, anthropogenic influences can either accentuate or reduce nitrification rates and can have a large impact on water quality.

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OCCURRENCE OF ORGANOCHLORINE PESTICIDES IN VEGETABLES GROWN ON UNTREATED SOILS FROM AN AGRICULTURAL WATERSHED

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INTRODUCTION

The role of agrochemicals in modern agriculture is continuously evolving, and their contribution to crop protection continues to increase. At a global level, the use of pest control agents (i.e., pesticides) has proved to assist the solving of many problems facing human health and food production. Organochlorine pesticides (OCPs) are a class of compounds of particular concern in the environment because of their recalcitrance in natural solids, global transport, distribution, and toxicity, which leads to include them into the group of persistent organic pollutants (1). This group had been widely used since early 1940, with an intensified usage during the Second World War. At that time, crops were duplicated and malaria incidents were reduced in India from 75 millions to less than 5 millions persons affected in 10 years as the result of DDT usage. However, such usage has occasionally been accompanied with hazards to man and the environment. Target species became resistant, leading to the application of higher doses of pesticides. For these reasons, most developed countries banned or restricted their usage since 1970. However, in the poorer countries, OCP restrictions or prohibitions date from the later 1990s. In addition, many of the banned or withdrawn pesticides from the developed market are still produced and sold in developing country market (2). Pesticides such as DDT are still in use in India and Brazil with public sanity purposes (3–5).

When pesticide are agriculturally applied, roughly 85–90% of pesticides never reach the target organism, but disperse through the air, soil, and water (6,7). Pesticides that are bound in soils or taken up by plants and animals are drawn into rivers and lakes and move into the aquatic food chains inducing severe damage to aquatic life (8,9). The environmental behavior of OCPs is largely related to their physical and chemical characteristics. Some of them are volatile enough to evaporate, reaching the upper atmosphere and being carried hundreds or

thousands of kilometers by a process known as “Global Distillation” (1,10,11). Thus, the capability of these pollutants for long-distance transfer, precipitation, and accumulation in environmental materials results in the formation of background concentrations in remote regions from the emission sources (12). Therefore, OCPs can be found in many places where they had never been applied. They have been linked to carcinogenicity and endocrine disruption in mammals, and concerns over toxicity are exacerbated by their hydrophobicity, which results in bioaccumulation in fatty tissues and biomagnification through food chains (13).

The general population can be exposed to low levels of pesticides in three general ways:

1. vector control for public health and other nonagricultural purposes
2. environmental residues
3. food residues

Regarding food residues, consumption of plants or plant products, such as vegetables, fruits, and grains, form a major part of the food consumption of human beings and cattle (14). Moreover, vegetables are an essential component of our diet because of their nutritional value (15).

The impact of soil contaminants on human health and accumulation of pesticides in food crops is a great concern because of increased dietary exposure of consumers (16).

Considering that vegetation is the link among the atmosphere, soil, and human food supply, contamination of plants will have a great influence on the total daily intake of a substance (14). Thus, it is important to understand those processes by which pollutants enter into this environmental compartment.

ABSORPTION AND ACCUMULATION OF OCPs IN PLANTS

Plants may accumulate OCPs via different pathways:

1. adsorption to the root surface
2. root uptake and transport to the upper plant tissues
3. uptake of airborne vapors by aerial plant parts (stem, leaves, and fruits)

The relative significance of each route by which OCPs can be taken up into fruits and vegetables is likely to be influenced by a wide range of factors (i.e., physicochemical characteristic of the pesticide, environmental conditions, and specie-specific characteristics) (17,18) (Fig. 1).

Air to Plant Route

OCPs are emitted to the atmosphere via various routes: by transfer to the airflow during the treatment of agricultural crops, evaporation from treated surfaces of plants and soils, and finally by wind erosion (12).

Simonich and Hites (19) presented a simplified mechanism of pollutants absorption by plants from air. Air to vegetation transfer of gaseous pollutants will be on the partition coefficient of the pollutant in the octanol

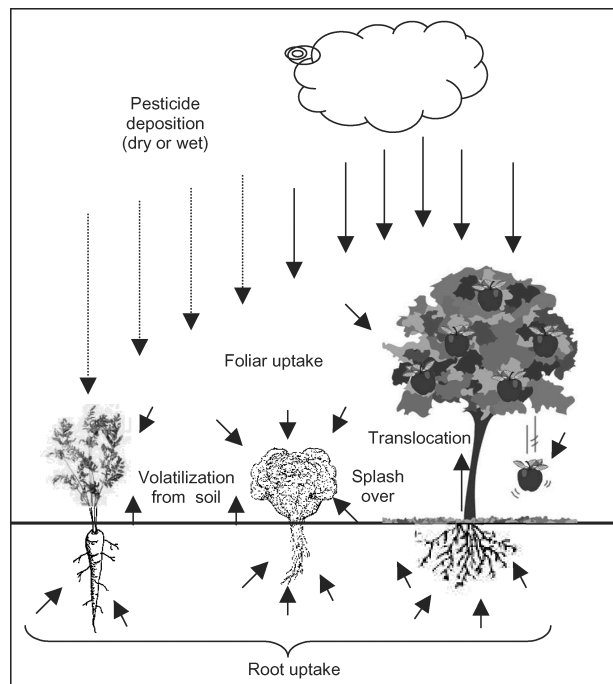


Figure 1. Pathways for accumulation of Organochlorine Pesticides by plants. Modified of Lovett et al. (18).

air system, the surface area, and lipid concentration in plant tissues.

Plant cuticles are the first barrier in the absorption of airborne and fall out OCPs in all aboveground organs and parts of higher terrestrial plants (fruit, flowers, stems, and leaves). Moreover, gaseous contaminants can directly reach the intercellular aerial medium through the open stomata (20–22). It was found that OCP residues in plant tissues decrease very slowly because of their low solubility in water and high solubility in oils. Thus, sorption of pesticides by plant surfaces depends significantly on the morphological features of the leaf, velocity, waxing, etc. (12,23). Moreover, vapor transfers of OCPs from soil to the aboveground parts of plants would be expected to occur to varying extents for all crops. For vegetables such as lettuce, courgettes, or zucchinis, which develop in contact with or in close proximity to the ground, it is possible that uptake of OCPs could also occur by direct adsorption from the soil and by splash-over of soil particles during heavy rain (12,23,24).

Soil to Plant Route

The transition of OCPs from soils into plants (35–70%) will be, in general, higher than into adjacent environments: water (2–18%) and atmosphere (18%) (12).

The uptake of OCPs by plant roots is affected by the bioavailability of the pesticide in the soil, leading to a competition between the organic fraction of the soil and the root material. Higher soil adsorption will result in a lower plant bioavailability (25).

After root uptake, plants can translocate pesticides via xylem to the aboveground plant parts. The extent of such mobilization will be on pesticide solubility and

plant characteristics, such as root exudates (26). Thus, pesticides whose *K_{ow}* are higher than 4 will not be able to translocate from root to aerial tissues in most plant species (17). However, an exception has been reported for vegetables from the cucumis and cucurbitacea genera, which are able to translocate highly hydrophobic compounds, such as the insecticide DDT, PCBs, and dioxinas, from roots to fruits (26–29).

OCPs OCCURRENCE IN VEGETABLES

As was previously mentioned, vegetables are essential components of the human diets. As pesticide residues remain on fruits and vegetables, it is important to know the magnitude of such contamination from the point of view of safety to consumers. Thus, a growing concern for safer foods has led research into increased pesticides residues monitoring.

In order to regulate and control such residues, the concept of maximum residues levels (MRL) was established. This level represents the maximum concentration of the residue (expressed in mg/Kg) that is legally permitted in specific food items and animal feeds (31).

OCP residues in vegetables are included in the Food and Drug Administration (FDA) monitoring program because they are still present in the environment (32). A residual content of these compounds in foods today is still quantifiable, and their avoidance in farming is largely impossible (33).

Regular surveys and monitoring programs of OCP residues have been carried out for decades in developed countries, such as the United States (32,34), the United Kingdom (35), and Spain (36). Although in some developing countries, such as India (37), Tunisia (38), Nigeria (39), Rumania (40), Brazil, the Philippines (41), Egypt (42–44), Honduras (45), and Argentina (46), reports about OCP levels in vegetables were made, they were not made as a part of governmental monitoring programs. Most of the reports showed that OCPs, such as HCHs, DDTs, Aldrins, Chlordane, and Heptachlor, were present in all analyzed samples. In the United States, about 1% of the samples violated the established MRL (32), but in developing countries, such as Brazil, this value can increase up to 3.7% (41). The presence of OCPs in vegetable samples is the consequence of either recent and current use in developing countries and/or the result of their persistent residues 30 years after their prohibition.

One alternative to synthetic pesticide use is organic farming. Thus, two decades ago, growing environmental awareness and concerns about safer foods have lead people to question modern agricultural practices, which has been reflected in an increasing demand for organic production, which is perceived as less damaging to the environment and to be healthier than conventionally grown foods (47,48).

ORGANIC AGRICULTURE: WORLD WIDE SITUATION

Organic agriculture has developed rapidly during the last decade, and governments have encouraged the

development of this tillage practice, mainly in developed countries. Chemophobia is the commonest reason for the public to choose organic food on the assumption that such food is free of synthetic pesticides (49).

Organic farms distinguish from all other forms of farming, namely conventional, by a rejection of soluble minerals as fertilizer and synthetic pesticides in favor of natural ones (49). However, organic food contains pesticide traces, although the amounts are lower than in conventional produce (50,51). Although it is likely that organically grown foods are lower in pesticide residues, very little documentation of residues levels exists (52). Moore et al. (33) studied the residue of eight OCPs in two matrices (carrots and winter squash) used to elaborate conventional and organic baby food brands. They found no detectable residues among all types of food. These results were the consequence of processing fruits and vegetables (washing, peeling, and cooking) that largely eliminates residues, as was largely demonstrated (53–55).

Today, developing countries represent a source of a wide range of organic produce, especially tea, coffee, cocoa, herbs, and spices, as well as tropical and subtropical fruits and vegetables for Europe and the United States (56).

Vegetable diversity produced in Argentina reaches 115 species with field crops covering a surface area of some 600,000 hectares (57), and about 160,000 hectares are destined to organic production. The European Union, EE.UU, and Switzerland are the main markets of destiny for these products (58). Therefore, the study of the factors that may impact in this kind of production is relevant. Most organic farms are supposed to remain small and self-sufficient, whereas conventional farms are increasing in size as the result of economic pressures.

The environmental fate of OCPs in agricultural environments of Argentina have been studied, including the impact of these pollutants in watersheds characterized by rural and small-case farming land use (59–62). OCPs were found in vegetables from organic and conventional farming (46,52,63,64). The presence of these contaminants in the organic farm was an interesting finding.

OCPs IN VEGETABLES ORGANICALLY CULTIVATED

Previous to the settlement of the organic farm, the area was characterized by natural vegetation. The farm is surrounded by important hedgerows and trees, and neighboring fields on both sides are conventionally farmed since its establishment about 30 years ago. Agrochemical use was a common practice in surroundings fields, and unsprayed field margins were not established.

From the organic farm, several vegetable species were studied in order to estimate the OCPs levels in their tissues. The produces include leafy vegetables (lettuce and chard), tomato, and leek. Most of the studied crops showed similar OCP levels between root and aboveground plan parts (stem or leaves). This observation was not consistent with previously published data where plants were cultivated in treated soils (28,30,65). Thus, in the organically treated soils, the air to plant route would be as important as the soil to plant route in all species.

The main OCPs found in vegetables were the same that were previously reported for conventional produce: HCHs, DDTs, and Chlordane being these pesticides forbidden for agricultural purposes. Moreover, high levels of endosulfans (α - and β -isomers and sulfate metabolite) were also found as a consequence of their current usage. All species accumulated OCPs efficiently from soil in their roots, leading to high root bioconcentration factors. Common practices in organic tillage constitute the enrichment of soil with vegetal rests. In this farm, pine needles collected from surrounding forests are usually added to the soil at the beginning of the growing periods. Pine needles, rich in waxes, are known to accumulate highly lipophilic pollutants from the atmosphere (66,67). Therefore, those soils enriched with this amendment showed the highest level of OCPs. Moreover, plants growing in such soils showed enrichment in the more lipophilic compounds in their roots.

Aerial tissues showed a great variation in accumulation regarding species and tissues. In lettuce, differences between inner and outer leaves were found, with the lowest concentrations in the outer leaves despite the higher lipid content, maybe by the growth dilution effect or weathering. Moreover, plant morphology could be an important factor affecting OCPs uptake and distribution (23,24). Thus, lettuce with outer leaves enlarging inner leaves had higher OCP levels in stem and inner leaves, probably as a consequence of the accumulation of pollutants between outer and inner leaves. On the other hand, chard plants with an inverse wide-high relation and longer stem showed higher OCP levels in outer leaves (51).

Tomato fruits contained OCPs not only in their peel but also in the inner flesh tissues, which indicate absorption rather than adsorption (63). Total OCPs in flesh were 3-fold greater than in peel, probably as a consequence of either continuous uptake by aerial tissues (fruit peel, leaves, and stem) of airborne OCPs, followed by translocation via phloem throughout the plant and accumulation in fruit flesh or removal of pesticides from peel by weather influences as photodegradation, volatilization, and so on (17).

Acaricide Dicofol is still in use in this country and has been reported to contain a significant amount of DDT as manufacturing impurity. Thus, it represents a potential continuous source of DDT in the studied area. DDT breaks down to DDE and DDD in soil and generally with time the parent/metabolite ratio decrease (68). DDT/DDE ratios found > 1 suggest that besides Dicofol use, new DDT can be continuously transported to the atmosphere of the organic farm. The presence of this new DDT can be attributed to illegal use or to atmospheric transport from those countries, mostly tropical, where DDT is still in use (61). Moreover, data suggest that long-range transport could be the predominant source of chlordane in the farm. The α/γ -HCH ratio can be used as diagnostic of HCH sources, inputs of γ -HCH (lindane) decrease the ratio found in air and vegetation containing a background of technical HCH ($\alpha/\gamma = 4-6$) (68). The low α/γ -HCH ratio found in all samples reflect recent inputs of lindane to the atmosphere of the studied area.

Endosulfan is one of the few organochlorine insecticides still in use; it is primarily used on summer crops such as tomatoes in the agricultural belt on the northwest area of the studied watershed. Occurrence of endosulfan in aerial tissues of tomato agrees with their summer usage in the surrounding farms, showing the relevance of drift pesticide during application events. Research has reported that drift of endosulfans from aerial agricultural spraying can produce lethal concentration to fish in shallow exposed water bodies 200 m away from the target spray area (8). In this watershed, it is also common during summer months that a massive "fish kill" can occur. One possible explanation could be that lethal concentration inputs of endosulfans applied in the surrounding area that reach the waterbody. Moreover, surface runoff from the organic farm may constitute a source of endosulfans to the waterbody.

As a whole, it was demonstrated that plants bioaccumulated OCPs efficiently from both routes, air and soil. In organic tillage, it is common that crop residues are reincorporated into soil after harvest. Thus, soil OCP values may reflect contributions from decomposed plant material rather than changes either within soil or via the atmospheric deposition. Therefore, plant residues constitute a way of entry of atmospheric pollutants into of the organic farm, leading to the subsequent availability for root uptake. This fact will be relevant for plants whose edible parts grow below ground (i.e., carrots, potatoes) or those that are able to translocate them. Despite OCPs being detected in all edible tissue (leaves, stem, and fruit in the studied vegetables), the levels were below the maximum residue limits (MRL) considered by both Código Alimentario Argentino (CAA) (69) and the Codex Alimentarius (CA) (31).

Despite the practice of organic farming management, OCPs were present in the soil environment and vegetable tissues confirming the importance of contamination from adjacent conventional farms. Therefore, the occurrence of OCP residues in the organic samples may be attributed to ecosystem factors such as pesticide runoff, drift from pesticide used by neighboring farming, volatilization, leaching, and washout. Thus, when organically grown vegetables are subject to such potential sources of contamination, the meaning of the term "organic" becomes obscure and is difficult to interpret especially from the perspectives of the public consumers.

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PESTICIDE CHEMISTRY IN THE ENVIRONMENT

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INTRODUCTION

Pesticides have greatly enhanced the production and quality of food, feed, and fiber, as well as the control of disease vectors and pests adversely affecting the health and welfare of the world. Specifically, pesticides have been used in: (a) agriculture to increase productivity and quality and quantity of food, feed, and fiber; (b) forestry for pest control in silviculture; (c) the industrial, commercial, municipal, and military sector for rodent and weed/brush control around plant sites and in right-of-ways; (d) medical vector control; and (e) urban environment for termite control around structures and pest control in gardens (1). As a consequence, of all the environmental contaminants, pesticides have probably been the most widely criticized because of their direct use in natural systems. The nature of pesticide usage often requires broadcast distribution over large areas of crops. This wide treatment of the crop environment often results in treatment of adjacent areas as well. The use of pesticides is indispensable for conventional labor and extensive farming systems. Although the use of pesticides has resulted in increased crop production and other benefits, it has raised concerns about potential adverse effects on the environment and human health.

ENVIRONMENTAL BEHAVIOR OF PESTICIDES

The proportion of applied pesticide reaching the target pest has been found to be less than 0.3%, thus leaving over 99% elsewhere in the environment (2). Environmental entry and transport processes can occur either between or during rainfall events, including wet and dry deposition, foliar washoff, and spray drift. Moreover, pesticide dissolved in atmospheric water vapor may be deposited either on the soil surface directly or indirectly to the foliar surface, with subsequent removal during rainfall events via the process of foliar washoff. Once in the soil, pesticides may volatilize

into the air, enter the surface and groundwater by runoff or leaching, be taken up by plants and soil organisms, or stay in the soil (3–13).

As the use of pesticides in agriculture inevitably leads to exposure of nontarget organisms (including humans), undesirable side effects may occur on some species, communities, or on ecosystems as a whole. Therefore, the knowledge of pesticide environmental behavior, like adsorption to soil, leaching to groundwater, and volatility in the atmosphere, is of primary concern for an accurate assessment of the risk to the environment and humans (14).

SORPTION TO SOIL COMPONENTS (ORGANIC AND INORGANIC)

The soil is a complex medium whose physical, chemical, and biological properties are highly heterogeneous. The horizontal and vertical distribution of textural and structural elements in soil, and the differences in physicochemical properties, lead to variations in sorption, desorption, transport, and permeability of soil. The soil serves as a temporary or permanent sink for any chemical substance that hits its surface (12).

The sorption of pesticide on soil reduces its mobility; it is dependent on the physical and chemical properties of the soil as well as the molecular characteristics of the pesticide. Soil organic matter is the main soil constituent responsible for sorption of nonionic pesticides. In order to assess pesticide mobility, a sorption constant, based on the soil organic carbon (K_{oc} , $\text{dm}^3 \text{kg}^{-1}$), can be used. Thus, $K_{oc} = K_d / oc$, where K_d is a measure of the extent of pesticide sorption by the soil and oc is the fraction of the organic carbon presents in the soil (15).

A significant proportion, typically ranging from 20% to 70%, of a pesticide or its metabolites may remain in the soil as “persistent residue,” sequestered in soil colloids (16).

The sequestration results in a diminution in the rate and extent of biodegradation by microorganisms in the field. The decline in toxicity and bioavailability as a result of the accompanying sequestration may represent a natural remediation process (17,18).

BIOLOGICAL AND CHEMICAL DEGRADATION

Losses of pesticide in the soil via microbiological and chemical pathways are collectively termed degradation. The rates of degradation can be characterized by a half-life (DT_{50}). In soil, this rate increases with temperature and water content (19). The products of pesticide degradation, named metabolites, may have environmentally undesirable characteristics. Therefore, when the evaluation of the environmental fate of a pesticide is carried out, it is recommended to take the fate of metabolites into consideration.

VOLATILIZATION

Volatilization is a major cause of pesticide loss from target areas, particularly when pesticides are applied to the

surfaces of soils or plants. Atmospheric transport and deposition are the major processes for distributing many pesticides around the world (20). The most appropriate criterion of the volatilization rate of a pesticide is Henry's law constant (K_h , dimensionless), the ratio of the vapor pressure to the water solubility (21,22). Compounds with K_h much greater than 2.5×10^{-5} are volatile, with volatility decreasing with time, whereas compounds with K_h much smaller than 2.5×10^{-5} are much less volatile, with volatility increasing with time (21–23).

RUNOFF AND LEACHING

Water may disperse pesticides into the environment via foliar washoff, surface runoff, and leaching. Runoff may contribute to pollution of surface water and leaching to contamination of groundwater.

Runoff is defined as the water and any dissolved or suspended matter it contains that leaves a plot, field, or small single-cover watershed in surface drainage (15). Runoff may include dissolved, suspended particulate, and sediment-adsorbed pesticides. Pesticides that remain at the soil surface for longer periods of time because they are strongly adsorbed and resistant to degradation and volatilization will be more susceptible to runoff, whereas incorporation into the soil will reduce runoff risk (24). Soluble pesticides may be more readily leached into the soil during the initial rainfall.

Pesticide leaching may cause groundwater pollution. The extent to which groundwater contamination occurs will depend on pesticides properties, soil characteristics, drainage rate, and water table depth. A combination of mobility and persistence determines whether a compound will be degraded during its residence time in the zone above the groundwater (21,25). Gustafson et al. (25) propose a single numerical index for predicting the water contamination potential of pesticides, the groundwater ubiquity score (GUS). It is defined as $GUS = \log(DT_{50}) \times (4 - \log(K_{oc}))$.

Pesticides detected in groundwater generally have GUS values exceeding 2.8, whereas compounds with GUS values below 1.8 were not detected in groundwater.

Rate of pesticide leaching in soil decreases with increasing organic matter content and depth of the surface zone with high biological activity. Moreover, the presence of macropores (cracks, worm holes, root channels) in soils enhances the hazard of pesticide leaching to the groundwater. Via these macropores, water and solutes can be transported rapidly to the subsoil and groundwater, bypassing much of the soil matrix (26).

PESTICIDES GROUPS

Over the last 20–25 years, a drastic change in the general characteristics of pesticides has occurred. They have changed from those with extremely low water solubility, strongly sorbed with long half-life to those that are more water soluble, only slightly sorbed, and less persistent. According to the chemical structure, pesticides can be classified as organochlorines, extremely

hydrophobic with very high partition coefficient, and as organophosphates, carbamates, and pyrethroids, which are more hydrophilic with lower partition coefficient value (27). Moreover, pesticides can be conveniently divided into classes, depending on which particular pest they are directed toward. Thus, the main groups are the herbicides, insecticides, and fungicides.

Pesticide properties, combined with the amount applied, determine how many pesticides can be transported along each route into the environment.

As was mentioned above, the low water solubility and the high partition coefficient of the organochlorine pesticides (OCP) confer to these pollutants a high environmental persistence. For this reason, the monitoring of these compounds is highly recommended.

WHAT ARE "ORGANOCHLORINE PESTICIDES"?

Organochlorine pesticides are important classes of environmental contaminants. Owing to their environmental persistence, toxic properties, and potential for bioaccumulation, they can be classified as "persistent organic pollutants" (POPs). Of additional concern is the ability of many POP compounds to undergo long-range atmospheric transport, developing from their semivolatile nature, which has resulted in their distribution in nearly all environmental compartments on a global scale, often far removed from their places of production and application (28–30).

OCP have been widely used throughout the world since the 1950s, although the use of many of these chemicals has been subjected to restriction and/or banned in many countries in recent years. However, the use of OCP in tropical and subtropical countries has been increasing in recent years in agricultural and public health programs (31). In addition to current usage, significant environmental burdens remain for many of these chemicals, which can act as secondary sources from which releases to the environment can occur, even decades after initial use (32).

OCP enter the soil by deposition from air, drift, or by washing off from plant surfaces during rainfall or irrigation. As a result of their strong hydrophobic (lipophilic) character, these compounds are mainly found associated with organic matter in soil and lipid tissues of organisms.

Organochlorine Pesticide Distribution in Relation to Land Uses: A Study Case

Knowledge of pesticide concentration and spatial distribution is necessary in order to assess the environmental impact of these compounds. A field study was carried out as part of an integral research about the dynamics of OCPs in a shallow lake watershed, Argentina, in relation to surrounding land uses (12).

The primary objective of this study was to assess the effect of land uses, such as agricultural, recreational (used for hiking, camping, and campfires), and natural (forested area that have never been cultivated) on the occurrence and distribution of organochlorine pesticides in soils from Argentina.

The organochlorine compounds analyzed included α -, β -, γ -, and δ -HCH; p,p'-DDT and its degradation products; p,p'-DDE and p,p'-DDD; heptachlor and its metabolite (heptachlor epoxide); aldrin; dieldrin; α - and γ -chlordane; and α - and β -endosulfan. Three depths were selected from each land use (0–15 cm, 15–30 cm, and 45–55 cm).

The results showed that, as was expected, the soil total organic carbon (TOC) decreased with depth for all three land uses. For the natural soils, total OCP concentrations were well correlated with the levels of TOC through the soil profile, whereas in agricultural and recreational soils, this correlation was not observed. These results imply that although TOC is known to be the most important OCP sorbent in soil surface horizons, additional factors exist, such as particle size characteristics and organic matter composition, and physicochemical characteristics of OCPs involved in pesticide retention.

Natural soil accumulated OCP to a greater extent than the other soils, even though it has never received direct OCP application. It is located on a hill that is clothed with trees and grass cover, which would act as an OCP trap, avoiding the OCP to leach up through soil profile, volatilization, and minimizing the risk of soil erosion. The highest concentrations were found in the surface layer, decreasing gradually with depth as a result of an undisturbed soil profile. DDT (472.5 ng/g dry wt), DDE (99.98 ng/g dry wt), and HCHs (25.44 ng/g dry wt) were the main pesticides found. Moreover, the levels of OCP in soils from an organic farm near to these natural soils ranged between 4–12 ng/g dry wt depending on the enrichment with pine needles collected from the surrounding natural areas (33,34). It is known that the waxy surfaces of pine needles has the ability to bioaccumulate airborne lipophilic pollutants (35). The soils from the organic farm could be considered as "natural soil," because it has never received direct pesticides application, either the lack of permanent vegetal cover and the soil disturbance by tillage practice lead to lower OCP levels. However, in both soils (natural and organic farm), a similar OCP pattern was found, as a consequence of the exposure to the same agricultural environment.

The values of natural soils are similar to those found by Dimond and Owen (36), (in Maine) for forest soils in sprayed areas during 1958 to 1967.

DDT was heavily used in Argentina, but it was totally banned in 1990. Commercial acaricide (Dicofol), still in use in Argentina, has been reported to contain a significant amount of DDT as manufacturing impurity and might represent a continuous source of DDT in the studied area. However, the high values of DDT found in these soils suggest that "new" DDT continues to be atmospherically transported from other countries, mostly tropical, where DDT is still in use (37–40).

High levels of heptachlor epoxide, a metabolite of heptachlor, were also found possibly because of physicochemical processes, such as photolysis and photodechlorination, and because of the abundance of microorganisms and edaphic biota in the surface layer of these natural soils, biotransformation processes are also assigned.

Chlordane residues were found in all the samples with the highest values being those from the surface layer

of natural soil. The ratios γ/α -chlordane found in these soils suggest direct application of technical chlordane in this zone.

Regarding to HCH group, the surface layer of the three studied soils showed higher concentration of lindane than the deeper depth, although there were not marked differences among depths. Lindane is one of the few OCPs analyzed in this work that is currently being used in Argentina.

Those soils dedicated to recreational activities showed the highest OCP values in the middle depth. During flooding periods, weathering of the surface layer occurs; as a result, a clearance of OCPs and predominance of HCHs are observed. The contribution of these compounds would come from the lagoon water; significant evidence exists that HCHs are the main pesticide group found in the aquatic biota of this waterbody (41).

In the agricultural soil during the last 25 years, a traditional tillage has been carried out in which a moldboard plough was used at a depth of 25–30 cm and in which straw was burned. This kind of tillage practice enhances organic carbon and fine particles (clay and silt) loss (30% in relation to natural soils), jointly OCP loss by volatilization or surface runoff. In addition to this management practice, the topography enhanced by slopes, wind, and water erosion during rain and irrigation periods justify these losses. Despite this, high values of OCPs in the surface layer were detected, with DDT being the main pesticide found, supporting the assumption that, because these pesticides (except lindane and endosulfan) are forbidden in Argentina, they would reach these soils via atmospheric transport.

Finally, soil characteristics, topographic position, and land uses justify the differences in OCP accumulation among the studied soils. The accumulation pattern represent a dynamic balance between deposition and erosion in association with wind-induced circulation, tillage management practices, and effects by natural processes, such as rainfall and flooding periods, according to different soils.

The results of this study suggest that natural areas should not be overlooked when assessing the sources and monitoring the occurrence of OCPs.

RELATION TO SURFACE WATER

The use of pesticides on agricultural land may have a negative effect on nearby aquatic ecosystems (12,13,42,43). This effect is determined by the specific chemicals used, the place and methods of application of the chemicals, and the various mechanisms available for transporting compounds to the waterbodies, with soil erosion being a major pathway (44). The extent to which pesticides move from treated soils depends on the physicochemical properties of the different active ingredients, the soil management practices, and the combination of environmental variables, such as frequency, intensity, and duration of precipitation (45).

Hydrophobicity and persistence are the main properties of pesticides that control their accumulation in sediment and aquatic biota. Thus, this fact confirms the current necessity of OCP monitoring.

CONCLUDING REMARKS

It is known that pesticides are released into the environment as formulated products that are designed to deliver the active ingredient to the target organisms. During the last forty years, a transition in the type of used pesticides has taken place. Thus, the highly persistent and bioaccumulatives organochlorine pesticides were replaced by organophosphates, carbamates, and pyrethroids, currently used, with faster degradation in the environment.

However, despite pesticides undoubtedly contributing to increase farm productivity and human health, they also create several problems including widespread accumulation of residues with damage to wildlife, fisheries, beneficial insects, and even humans as a result of their dispersion toward nontarget areas. Today, the concern is the impact of those “environmental healthy pesticides,” because they are being linked to a high fish, wildlife, and bees toxicity, and some of them, because of their high water solubility, are potential groundwater contaminants. For these reasons, the monitoring of pesticide applications is necessary because it can be helpful in modifying the application techniques or use of particular chemicals to avoid adverse effects. Concern over possible impacts of pesticides on the environment and humans is not confined just to the highly industrialized nations. Developing countries are also experiencing the same concern exacerbated by lack of infrastructure and resources to deal with them.

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REMEDIATION OF PESTICIDE-CONTAMINATED SOIL AT AGRICHEMICAL FACILITIES

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AGRICHEMICAL FACILITIES

An agrichemical or crop production facility is a commercial agricultural application business. These facilities store, sell, mix, load, and apply pesticides and fertilizers to fields for farmers and livestock producers. There are about 9,250 of these relatively small businesses in the United States and about 850 registered agrichemical businesses in Illinois. Pesticide mixing, loading, and handling sites are diverse and vary in the types of products handled at the site, the size of the facility, and the construction and type of containment (1).

Sources of Water and Soil Contamination

Bulk liquid and dry pesticides and fertilizers are typically mixed, repackaged, or transferred from one container to another. Accidental and incidental chemical releases such as spills, tank leaks, hose breaks, and transport accidents can result in contaminated water and soil at agrichemical facilities. Moreover, catastrophic events such as fires and floods can also result in soil, sediment, and water contamination.

Extent of Water and Soil Contamination

About 25% of the groundwater samples collected from 52 agrichemical facilities in Illinois contained 11 pesticides (2). The four most frequently occurring pesticides were bentazon, atrazine, trifluralin, and metolachlor. Nitrate was detected in 16 samples at concentrations ranging from 0.17 to 59 mg N/L. In additional sampling at four agrichemical facilities, 17 contaminants (nitrate, pesticides, and pesticide degradates) were detected. Atrazine, for example, detected in 15 of 22 samples, ranged from 0.54 to 61 µg/L. Environmental property audits conducted at 49 randomly selected agrichemical facilities in Illinois revealed that pesticides were detected in at least one soil sample collected at every site and at all sampling depths (3). The most frequently detected pesticides were alachlor, atrazine, metolachlor, trifluralin, pendimethaline, cyanazine, metibuzin, metribuzin DA, butylate, and α-BHC. Pesticides were detected to depths of 4.5 m. About 50% of all pesticide detections, however, were found in the gravel layer that serves as the parking lot/road base for each facility. In an extreme case, the gravel layer at an abandoned agrichemical facility in Illinois contained 4.4% trifluralin by weight (4). Pesticide-contaminated soil and groundwater at retail agrichemical facilities have also been documented in Minnesota, California, Florida, Michigan, Wisconsin (1), and Arkansas (5).

Movement of Pesticides and Fertilizers

A portion of the pesticides and fertilizers in soil may leach into groundwater. Many of the commonly used

pesticides and fertilizers are fairly mobile and can be leached as surface water infiltrates into the ground. The contaminated gravel, in which the bulk of pesticides may occur, poses a risk to groundwater quality (4). Hence, contaminated soil and gravel can adversely impact groundwater and possibly drinking water in rural communities. Because of this potential, it may be necessary to remediate contaminated soil to minimize its impact on water quality.

CLEANUP OBJECTIVES

Groundwater Cleanup Objectives

The values for groundwater-cleanup objectives depend on the specific chemical and the type of groundwater at the site. For example, in Illinois, Class I groundwater is called Potable Resource Groundwater, and Class II groundwater is General Resource Groundwater (6). A major distinction between the two types of groundwater is the hydraulic conductivity of water-bearing geologic materials: groundwater in an aquifer that has hydraulic conductivity equal to or greater than 1×10^{-4} cm/s is classified as Class I Groundwater. Therefore, Class II groundwater would be expected in less permeable materials. The groundwater quality standards for Class II groundwater are less stringent than those for Class I groundwater. Class II groundwater cleanup objectives are five times greater than those given for Class I groundwater. For example, the objective for atrazine in Class I groundwater is 3 µg/L, whereas it is 15 µg/L for Class II groundwater (6).

Soil Cleanup Objectives

The remediation of contaminated soil also requires establishing some type of cleanup goals or objectives. Such objectives are used to determine whether the soil is "contaminated" in a regulatory context. Depending on the specific type, agrichemicals are subject to various physicochemical and biological processes that can reduce their concentrations in soil and groundwater. Unlike other common contaminants such as chlorinated hydrocarbons, agrichemicals are *intentionally* released into the environment to promote plant growth and to limit the spread of weeds. A mixture of application-rate equivalents and risk-based processes to derive soil cleanup objectives have been promulgated in the Midwest. For example, the Illinois Department of Agriculture provides a default soil-cleanup objective (SCO) for surface soil (0 to 12 inches) and subsurface soil (12 to 60 inches) for each major pesticide used in Illinois. A facility also has the option of developing site-specific SCOs. These SCOs were derived from the application of soil screening levels (SSLs) developed by the U.S. Environmental Protection Agency (7,8). The resulting SCOs take into account the movement and chemical fate of a given chemical in soil–water systems. In the application of the SSL concepts, three physicochemical-fate mechanisms are considered: dilution in groundwater, adsorption by soil organic matter, and volatilization. Degradation by biotic or abiotic mechanisms is not considered. The SSL model provides

a dynamic link between groundwater concentrations and soil concentrations. The default subsurface (12 to 60 inches depth) SCOs were calculated using Class I groundwater quality standards. The default surface SCOs were also based on Class I groundwater standards with the exception of acetochlor, alachlor, aldicarb, atrazine, cyanazine, dimethoate, disulfoton, phorate, simazine, and terbufos. For these pesticides, application-rate equivalents are used as SCOs. As given in (9), each SCO was based on pesticide label rates to the upper foot of soil or of a soil/gravel mixture.

EXCAVATION OF CONTAMINATED SOIL

Once the SCOs have been established, the areas and depths of contamination by agrichemicals at an agrichemical facility can be estimated. The delineation of the area and depth of material to be removed is not an exact science. In practice, the boundaries for excavation are subjectively interpolated between the locations where soil cores were collected. The boundaries for excavation may also, in part, be determined by any physical constraints such as facility buildings, railroad tracks, or other permanent structures. Once the contaminated materials have been excavated from the dealership, they must be disposed in an environmentally acceptable manner. There are three options available: land application or spreading, landfilling, and above-ground treatment.

OPTIONS FOR REMEDIATION

Land Application

Land application for the disposal of soils contaminated with agrichemicals is a common approach in the Midwest and may be the least expensive and most readily available disposal technique (10). It is based on the premise that if contaminated soil is spread in a thin layer on an agricultural field, the pesticides and/or fertilizers will dissipate to smaller and harmless concentrations. In practice, the pesticides in the contaminated material are applied at the application-rate equivalent that would be appropriate for that field and would be expected to biodegrade with time. If more than one pesticide or fertilizer is present, the application rate is based on the most limiting rate for the chemical detected in the contaminated soil sediment, or fill.

When pesticide-containing soil is land applied, it may be necessary to minimize the amount of coarse material in the soil before application. Attempting to spread soil that contains a significant amount of coarse material such as gravel can damage most spreading equipment and be objectionable to the landowner or farmer. A practical technique to remove the coarse material is to use field-scale screeners. The excavated, unsorted material is fed into some type of mechanical screener to separate the coarse debris from the finer soil. Depending on the size of the equipment and the mesh size of the screen, a screener can process between 50 to 150 tons per hour. The coarse material may be used as a road base, field roads, or placed back in the original excavation. The fine

material is then land applied using spreading equipment. There is a relatively narrow window of opportunity in which site remediation and landspreading can occur: postharvest to preplanting. During the planting season, agrichemical facilities are in their busiest periods and are reluctant to allow any site excavation because it can hinder the operation of the facility and service to their customers. During the winter, it is advisable to avoid freezing conditions because the soil cannot be handled effectively if frozen.

Often, there is not enough room at an agrichemical facility for soil screening. In actual practice, the excavated soil is transported from the facility to an area where the soil can be screened and land applied. Compensation to local farmers or land owners for the use of land to establish a staging area for the screening process and for the agricultural fields can add to the costs of remediation.

Landfilling

Historically, pesticide-contaminated soil excavated at agrichemical facilities has been placed in landfills. When land application is not possible, using contaminated soil as a refuse cover or backfill may be an option. However, it may be more expensive to place the soil in a landfill. For example, if transportation costs are ignored, the average cost of landfilling pesticide-contaminated soil in the Midwest is about \$25 per cubic yard compared with an average of \$10 per cubic yard for land spreading. Moreover, limitations in landfill capacity may be a barrier to the disposal of contaminated soil in some areas.

Soil Washing

Although land application and landfilling are commonly used approaches for pesticide-contaminated soil at agrichemical facilities, other options are under study. It has been suggested that soil washing could be applied for a relatively wide range of pesticides (11). Soil washing is a treatment process in which contaminants are extracted from the excavated material using a solution containing chemical surfactants. For example, under laboratory conditions, a 3% solution of the nonionic surfactant polyoxyethylene-4-lauryl ether was efficacious for removing atrazine from contaminated fill samples that were collected at an agrichemical facility (12). After 1 hour of mixing the fill-liquid suspensions, about 85% of the atrazine was extracted, compared with 26% removal by deionized water.

Bioremediation

Bioremediation is another option for remediating contaminated soil. Biodegradation is a process in which microorganisms transform or alter the structure and properties of chemical compounds by enzymatic biochemical reactions. Biochemical transformations can break down toxic, anthropogenic compounds into less toxic metabolites, and/or into inorganic compounds. Bioremediation is the application of biodegradation to reduce the concentrations of contaminants in soil by enhancing the physicochemical conditions for degradation, the creation of optimum conditions for remediating contaminated sites.

For example, the biodegradation of trifluralin, metolachlor, and pendimethaline in gravel fill samples was promoted by mixing them with yard-waste compost under laboratory conditions (13). The initial concentration of trifluralin (2200 $\mu\text{g/kg}$) was reduced to about 505 $\mu\text{g/kg}$ in a 1:1 fill:compost mixture after 40 days.

The basis for enhanced biodegradation is creating an environment where microbes are more numerous thereby increasing the probability of significant degradation. There are several critical environmental conditions that affect microbial activity: moisture content, oxygen, redox potential, pH, nutrients, temperature, and especially the supply of nitrogen and phosphorous. If the contaminated soil contains a large amount of carbon but little nitrogen and phosphorous, biodegradation will cease when available N and P are depleted. Therefore, fertilization of the contaminated fill materials may be required as a management technique to enhance biodegradation. For example, the addition of cornmeal can enhance the extent of biodegradation of pesticide-contaminated soil from an agrichemical facility when the material is land applied (14).

The addition of a mixture of cornmeal, surfactant, fertilizer, and clean soil, under laboratory conditions, increased the rate of biodegradation of atrazine, alachlor, and metolachlor in gravel fill samples that were collected at three different agrichemical facilities (15). The fill samples were then either moistened with sterilized deionized water to 25% moisture content, or amended with the surfactant-fertilizer solution containing 0.1% oleic acid, urea, polyoxyethylene-12-octyl phosphate (modified from the oleophilic fertilizer Inipol EAP 22). After 8 weeks of incubation, the amount of atrazine recovered decreased by 55 to 74% in fill-soil samples amended with cornmeal and the surfactant-fertilizer mixture. Similar reductions of alachlor and metolachlor were also observed in those samples (Fig. 1). No significant degradation of atrazine, alachlor, and metolachlor was observed in the unamended and sterilized fill samples. During the same incubation period, however, the rate of reduction of atrazine, alachlor, and metolachlor was slower in samples amended with cornmeal alone. The biodegradation half-lives of atrazine in the fill samples amended with cornmeal and the surfactant-fertilizer mixture ranged from 3.9 to 6.8 weeks.

Phytoremediation

Phytoremediation is a technique in which plants, primarily hybrid poplar trees, legumes, and grasses, are planted in contaminated soil. The plants have the potential to take up and incorporate contaminants into plant tissues, enhance biodegradation in the root zone, and reduce the volume of groundwater available to transport contaminants off-site. There are few, however, published examples of field-scale applications of phytoremediation at agrichemical facilities. Poplar trees were planted at an agrichemical facility in Illinois to remove nitrogen and herbicides from shallow surface groundwater (Eric Aitchison, Ecolotree, Inc., personal communication). A total of 440 trees planted grew to a height of 15 feet in 17 months, and a significant volume of groundwater was apparently used by the trees. Phytoremediation may have a future,

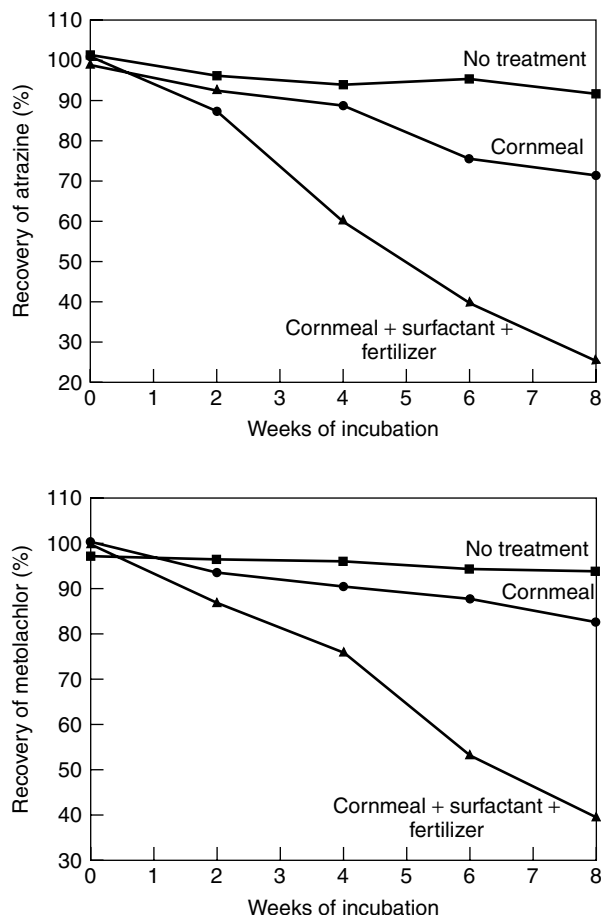


Figure 1. Amount of atrazine (top) and metolachlor (bottom) in soil-fill samples and the control sample (no treatment) in the presence of ground cornmeal and the cornmeal-soil-surfactant-fertilizer combination as a function of incubation time (15).

but additional research and development are needed. Phytoremediation, like bioremediation, may be too slow when rapid remediation is required. Phytoremediation may be suitable for inactive or abandoned agrichemical facilities. Moreover, relatively large concentrations of phytotoxic chemicals in soil and groundwater can limit plant growth (16).

OTHER APPROACHES

Other approaches for remediating pesticide-contaminated soil have been proposed such as incineration, solidification/stabilization, thermal desorption, radio-frequency heating, supercritical carbon dioxide extraction, and dechlorination by zero-valent iron (10,11,17,18). However, these approaches may not be cost-effective for a typical agrichemical facility. For example, incineration of soil may cost from \$800 to \$3,000 per cubic yard (17). However, if the contaminated soil also contains chemicals that are defined as hazardous under the Resource Conservation and Recovery Act, the soil may have to be incinerated (1).

THE SELECTION OF THE REMEDIAL APPROACH

Past handling practices have resulted in pesticide-contaminated soil and groundwater at retail agrichemical facilities. During about the last ten years, however, new containment regulations have been implemented in the Midwest which will hopefully reduce incidental releases of pesticides and fertilizers. For example, the Secondary Containment, Operational Area Containment, and Containment Management and Operations in Illinois (19) require the use of a containment system designed to intercept, retain, and recover operational and accidental spillage, leakage, wash water, and agrichemical residues. When contaminated soil, fill, or sediment is detected at an agrichemical facility during a site assessment, however, remediation may be required to minimize the adverse impact of the material on surface and groundwater. Techniques for remediating pesticide-contaminated soil is a current area of research, and new techniques and technologies are currently under study. However, land application and landfilling may be the most cost-effective, routine, and available technique to remediate contaminated soil at retail agrichemical facilities. Experience gained in the Midwest may be very useful for other domestic and international facilities.

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PESTICIDE OCCURRENCE AND DISTRIBUTION IN RELATION TO USE

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GENERAL RELATIONSHIP BETWEEN PESTICIDE OCCURRENCE AND USE

In the United States, about 1 billion pounds of pesticides are used each year to control weeds, insects, and other organisms and about 80% of them are used in agriculture (1). Widespread use of pesticides over the past several decades has led to their occurrence in groundwater, surface water, aquatic biota and sediment, and the atmosphere (2). In 8200 sample analyses conducted by the National Water Quality Assessment (NAWQA) program during 1992–1996, 58 pesticides were detected at least once at or above 0.01 µg/L in both surface and groundwaters in 20 of the nation's major hydrologic basins (3). The NAWQA studies also revealed that more than 90% of water and fish samples from all streams contained one or, more often, several pesticides

and about 50% of the sampled wells contained one or more pesticides; the highest detection frequencies were in shallow groundwater. In particular, the top 15 pesticide compounds found in water are among those with the highest current use, although some organochlorine compounds (e.g., DDT) heavily used in the 1960s were found most often in fish and sediments. The most frequently used pesticides occurred in geographic and seasonal patterns that mainly correspond to distributions of land use and associated pesticide use (4). Clearly, a significant relation can be expected between the use of pesticides and their occurrence in both surface and groundwaters. However, the extent of their occurrence and spatial and temporal distributions in surface and groundwater (how, when, and where the pesticides occur) may vary significantly.

ENVIRONMENTAL PATHWAYS OF PESTICIDES VERSUS THEIR OCCURRENCE AND DISTRIBUTION

The environmental pathways of pesticides determine their occurrence and distribution. Following release into the environment, pesticides can be washed off the target application fields into adjacent surface waterbodies (e.g., streams, rivers, lakes) by runoff and erosion, leached by percolating water down through the vadose zone into the underlying aquifers, volatilized into the atmosphere, taken up by plants, and accumulated in aquatic organisms through the food chain and even in human bodies. Thus, pesticides follow complex pathways and patterns of occurrence and distribution in the environment (Fig. 1).

TRANSPORT AND TRANSFORMATION PROCESSES OF PESTICIDES VERSUS THEIR OCCURRENCE AND DISTRIBUTION

Pesticides, existing in three phases (dissolved, adsorbed, and vapor phases) in the environment, undergo a series of physical, chemical, and biological transport and transformation processes such as advection, dispersion, degradation, sorption, volatilization, and partitioning between dissolved and vapor phases in surface and groundwater (Fig. 1). These essential transport and transformation processes dominate pesticide occurrence and distribution in the environment. Due to the complexity of these processes, the occurrence and distribution of pesticides in surface and groundwater can be extremely variable and are affected by numerous factors such as pesticide use, pesticide properties, land use, soil types, topographic features, agricultural practices, climatic conditions, hydrology, and hydrogeologic settings. Whether or not pesticides occur in a certain part of the environment (a specific river reach, soil layer, or aquifer) depends on three essential elements: (1) availability of pesticide, (2) availability of flowing water, and (3) accessibility/pathways.

VARIABILITY IN THE RELATIONSHIP BETWEEN PESTICIDE OCCURRENCE AND USE

The relationship between pesticide occurrence and use varies for different pesticides and geographic regions.

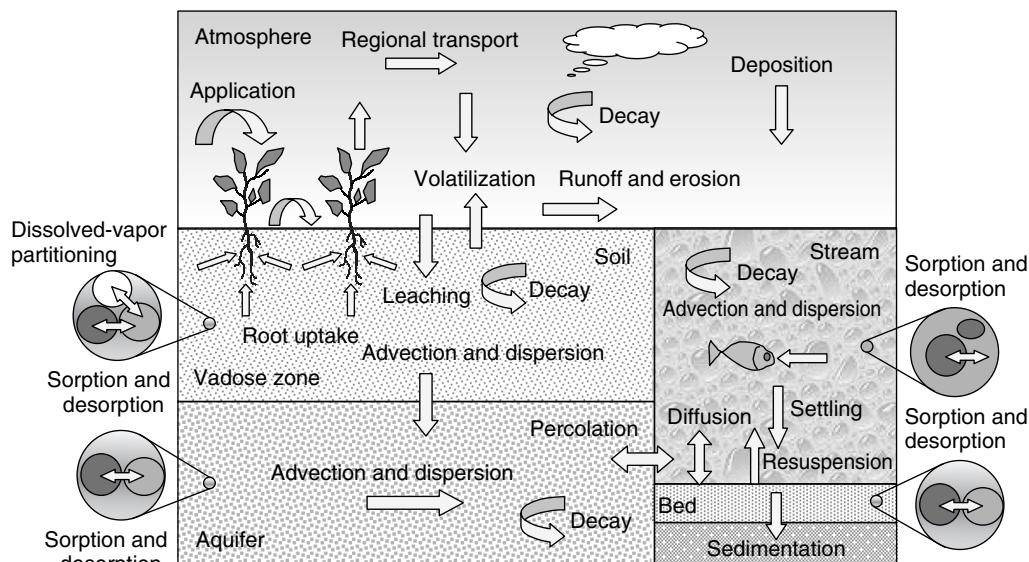


Figure 1. Fate and transport of pesticides and their occurrence in the environment.

Even for a specific pesticide at one site, the relationship changes with time due to the variability and complexity of hydrology. In addition, the relationship for surface water is different from that for groundwater due to their distinct hydrologic characteristics and pesticide fate and transport processes. Surface and groundwater systems have dissimilar timing of pesticide occurrence, temporal and spatial distributions, and magnitude of pesticide exposure levels. A pesticide application may result in a peak concentration in streams and shallow soils in hours, whereas it may take days, even months, to reach a peak exposure level in a groundwater system. The peak concentrations in groundwater can also be much smaller than those in surface water and shallow soils. A time shift of arrival of peaks can often be observed in the transport direction in both surface and subsurface water systems.

FIVE KEY PESTICIDE-USE ISSUES RELATED TO OCCURRENCE AND DISTRIBUTION

In terms of pesticide use, five key issues that affect pesticide occurrence and distribution need to be addressed: (1) which pesticide is applied (type and properties of pesticide), (2) how much and how often the pesticide is applied (amount and frequency of application), (3) when the pesticide is applied (timing of application), (4) how the pesticide is applied (method of application), and (5) where the pesticide is applied (domain of application). The types of pesticides that possess specific physical, chemical, and biological properties affect their occurrence and distribution directly by altering their environmental persistence, mobility, and other fundamental transport processes. The amount of applied pesticides determines the overall level of pesticides potentially available for occurrence in surface and groundwater, and the application frequency may determine the possibility and extent of pesticide buildup. The timing and method of application of pesticides affect occurrence primarily by changing their environmental

pathways and thus spatial and temporal distributions and exposure levels. Additionally, the overall patterns of pesticide occurrence and distribution vary with pesticide-treated crops (application domain) that have different growing periods and canopy characteristics and require different soil–water–pesticide management practices.

EFFECTS OF USE OF PESTICIDES ON THEIR OCCURRENCE AND DISTRIBUTION

Effects of Pesticide Properties on its Occurrence and Distribution

The capabilities of sorption and volatilization of pesticides, characterized by distribution coefficient and Henry's law constant, determine their existing forms (dissolved, adsorbed, or vapor phases) and thus their mobility. These two processes and the degradation of pesticides are critical to their occurrence and distribution. Pesticides of lower sorptive capability tend to move quickly with flowing water. The spread of this type of pesticide throughout the environment is, to a great extent, determined by water movement. In contrast, long-lived pesticides of strong sorption, such as DDT, may last decades in the environment (4).

Effects of Application Quantity and Frequency on Pesticide Occurrence and Distribution

Frequent applications of pesticides increase the total amount of pesticides released into the environment and hence endanger the vulnerable hydrologic system. In particular, the accumulative effect of long-lived and strongly sorbed pesticides is often of great concern.

Effects of Application Timing on Pesticide Occurrence and Distribution

The combined timing of pesticide application and rain-fall/irrigation can dominate exposure levels of some

organophosphate insecticides in both surface and sub-surface environments (5). A pesticide application, immediately followed by rainfall or overcanopy irrigation, provides an extremely high potential for the pesticide to transport from target fields to nearby surface waterbodies with runoff and erosion and to leach into soils and the underlying groundwater.

Effects of Application Methods on Pesticide Occurrence and Distribution

Pesticides can be applied by different methods, such as over-canopy spray, under-canopy ground surface spray, and soil-incorporated application. Application methods of pesticides affect their occurrence in the environment by changing their initial distribution and hence altering their following transport–transformation pathways and environmental fate. In addition, irrigation methods (e.g., over-canopy and under-canopy irrigation) and their combination with various pesticide application methods can be critical to pesticide occurrence and distribution. High variability of concentrations of some pesticides during irrigation periods can be attributed to differences in irrigation frequency and water management (6).

SUMMARY

The occurrence and distribution of pesticides in the environment are closely related to their use and are determined by a series of pesticide transport and transformation processes. The occurrence and use of pesticides follow extremely complex and dynamic patterns that are affected by numerous factors related to pesticide use, hydrology, and agricultural activities. Different ways of pesticide use (application quantity, frequency, timing, and method) can lead to distinct environmental fates of pesticides in terms of their occurrence, extent, as well as spatial and temporal distributions.

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ASSESSMENT OF POLLUTION OUTFLOW FROM LARGE AGRICULTURAL AREAS

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The pollution from large agricultural areas in general is discussed. The potential of remote sensing and geographical information system as input for NPS pollution assessment is emphasized. Mathematical models commonly used for assessing of nonpoint source (NPS) pollution are also described. Assessment of nutrient pollution from three agricultural watersheds of a typical Indian river basin ranging from 2–5 km² is considered as an illustrative example.

INTRODUCTION

The sources of surface water pollution typically fall into one of two categories: point-source pollution and nonpoint-source pollution. The term point-source pollution refers to pollutants discharged from one discrete location or point, such as an industry or municipal wastewater treatment plant, to surface water resources. The term nonpoint-source pollution refers to the discharge of pollutants, which cannot be identified as coming from one discrete location or point. The pollution outflow from agricultural areas is known as nonpoint source (NPS) pollution or diffuse pollution. NPS pollution occurs when rainfall, snowmelt, or irrigation water runs over land or through the ground, picks up pollutants, and deposits them in rivers, lakes, and coastal waters or introduces them into the groundwater. Sediment, nutrients, and pesticides are common agricultural pollutants, which cause surface water and groundwater quality problems (1–3).

Estimation of NPS pollution from cropland runoff is an important feature of regional water quality planning. The primary objective of NPS pollution estimation is to protect or restore the designated use of a water resource by reducing pollutant delivery to the water resource. All nonpoint sources of pollution are not equal. Many nonpoint sources of pollution are insignificant, whereas other sources contribute substantially to water resource impairment. Topographic, hydrologic, and agronomic factors often combine to make some nonpoint sources more detrimental to the beneficial use of water resources than others. Therefore, a method or strategy to identify and prioritize for treatment NPS areas, which are more detrimental than others, is desirable. A land treatment strategy should be developed to guide the selection and implementation of best management practices (BMPs). The BMP approach involves identifying and implementing land use practices in agricultural areas that prevent

or reduce nonpoint pollution. In the case of erosion-sedimentation, many such practices are well known for agricultural activities. However, for some other types of pollutants, the BMP may not be known. In such instances, research that relates land use to water quality is needed.

ROLE OF GEOGRAPHICAL INFORMATION SYSTEM (GIS) AND REMOTE SENSING FOR NPS POLLUTION ASSESSMENT

Like any environmental phenomena, there is a spatial dimension to the management of water quality and the control of nonpoint-source pollution in agricultural watersheds. Understanding the spatial relationships among the various pollution sources in a watershed is critical to the successful implementation of the best agricultural management practices. A geographical information system (GIS) enables the effective integration, management, and analysis of disparate data sets related to chemicals, soils, climate, topography, land cover, and land use. These data sets are important driving variables for a nonpoint pollution model (4).

Several strategies for integrating distributed water quality models and GIS have emerged during the past few years (5–7). In most applications, the GIS was used primarily to generate model input data and to display output data from the model—an approach usually referred to as loose coupling of a model and the GIS (8–10). In loose coupling, there are two options: (1) loose coupling through interchange of data files in ASCII format between the model and the GIS and (2) loose coupling using a common binary file (10). An interface program (e.g., pre- and postprocessor) is normally used to convert and organize the GIS data in the form that the model requires. The loose coupling can often be time-consuming and problematic, particularly, for large watersheds. The loose coupling also, depends on the data structure of the GIS and the data file format specified by the water quality model (11,12).

The second strategy of coupling the GIS and water quality models involves close coupling in which the control programs in the GIS software are slightly modified to provide an enhanced environment for data transfer between the model and the GIS database (8–10). The options for loose and close coupling have significant overlap, depending on the characteristics of the water quality model. However, in close coupling, information is passed between the model and the GIS via memory-resident data models rather than external files. This improves the interactive capabilities and performance of closely coupled modeling environments.

The third strategy of coupling water quality models and the GIS is tight coupling or full integration (8–10). This strategy is based on incorporating the functional components of one system (e.g., the model) within the other, thereby eliminating the use of interface programs. The most widely used option of tight coupling involves developing tightly coupled seamless interfaces between the model and the GIS. The major benefits, of tight coupling strategy include improved performance, enhanced user interface, and increased problem-solving

capabilities through efficient analysis of “what if?” scenarios. There is also little or no redundancy in the development process because the systems can leverage off each other.

Data acquired from remote sensors on aircraft and satellites offer great promise in providing numerous model inputs. Water quality models used to determine nonpoint source pollution vary in type and extent of land use for which these are best suited. Water bodies, urban areas, forests, cropland, and bare soils or rock can generally be differentiated very well. Individual species of vegetation often can be separated, surface soil properties can be delineated, and the percentage of vegetation or residue cover on the land can sometimes be estimated. The spectral, spatial, and temporal characteristics of remotely sensed data make those data useful for models that require knowledge of land use, cover, and conditions when calculating nonpoint-source pollutant loads. The relevant studies are available in many textbooks and the literature.

MATHEMATICAL MODELS FOR ESTIMATING POLLUTANTS FROM AGRICULTURAL AREAS

Mathematical models have been used extensively since the late 1960s to estimate pollutant outflows from agricultural areas. The term mathematical model may be new, but all of us have used mathematical models since we took algebra in school. Such models are mathematical representations of physical, chemical, biological, social, economic, and related processes. The advancement in computers, has made it possible now to evolve different mathematical approaches (from simple linear regression to complex differential equations) for obtaining favorable outcomes. Some of the mathematical models used for NPS pollution estimation are as follows: (1) SWMM (13), (2) HSPF (14), (3) ANSWERS (15), (4) CREAMS (16), (5) SWAM (17), (6) NTRM (18), (7) SWRRB (19), (8) SHE (20), (9) AGNPS (21), (10) SWAT (22), and (11) ANSWER-2000 (23).

Most of the mathematical models cited before simulate hydrologic, chemical, and physical processes involved in the transport of sediment, nutrients, and pesticides. In general, mathematical models for estimating NPS are divided into two classes: (1) a temporal class that includes event simulation and continuous simulation and (2) a spatial class that includes lumped parameters and distributed parameters.

In the temporal class, the continuous simulation models are preferred because they simulate all NPS pollution producing events and can be used to estimate total maximum daily loads and seasonal variations in NPS loading. However, in the spatial class, distributed parameter models are preferred. The principal advantage of distributed parameter models over lumped parameter models is that distributed parameter models represent the spatial variability of watershed features (soil, topography, land use etc.) more accurately. Moreover, the distributed parameter models can also estimate pollutant losses at different locations within the agricultural area. Some of the most commonly used distributed parameter

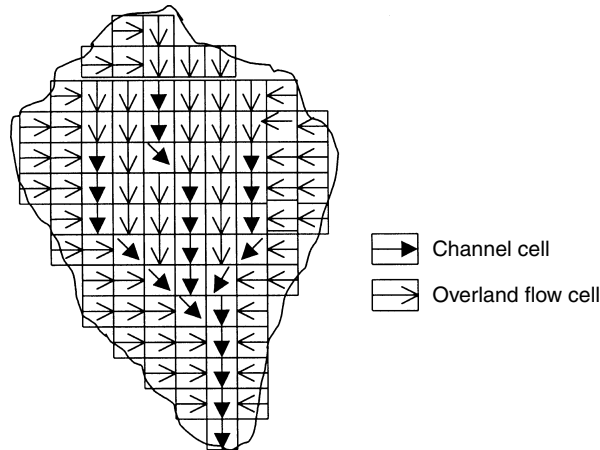


Figure 1. Mixed cell and routing process in overland and channel flow.

continuous simulation models are AGNPS and ANSWER-2000. The processes involved in these models are explained next.

In the beginning of the modeling process, a large agricultural area is divided into smaller grids or cells. A typical arrangement of such cells is shown in Fig. 1. The flow, sediments, and nutrients available at the beginning of the cell, coupled with those generated within the cell, are used to assess these values at the outlet of the cell. Subsequent routing of these values can lead to the estimation of flow as well as the NPS pollution load at the outlet of the agricultural watershed.

In a hydrological process, the rainfall added to an agricultural area in the beginning is intercepted by the canopy cover, evapotranspired or infiltrated. Infiltration capacity is very high initially because of movement of water to the unsaturated zone, and water within this layer percolates downward. Water is discharged to the channel drainage system, when infiltration capacity is exceeded by the water supply and the depression storage has been satisfied. The total inflow to the channel drainage system is a combination of surface runoff, interflow, and base flow, which is calculated by using a water depletion function to gradually diminish it. A few specific relevant equations for the computation of different flow components are shown in Appendix 1. However, many more equations can be obtained from the literature.

The sediment transport capacity and potential sediment are important variables of nonpoint source pollution from a large agricultural area. As seen from Appendix 1, the sediment transport capacity is based on a sediment concentration ratio, c , estimated with a shear-stress relationship between the dominant flow shear on the soil and the critical stress based on the Shields criteria (24). The potential sediment supply due to rainfall can be calculated from an empirical relationship for rainfall energy rate adapted from the Universal Soil Loss Equation (USLE) method. The method accounts for the effects of canopy cover by introducing cover factors into the calculations of soil detachment due to rainfall and runoff. Similar models available in the literature can also be applied for

estimating the sediment transport capacity and potential sediment.

The mathematical approaches used for the NPS water quality component in this chapter are widely used NPS models (Appendix 2). These were developed for the CREAMS model (25). Young et al. (20) modified the algorithms applied in the CREAMS model and created the AGNPS model, suitable on watershed scale. The nutrient simulation is normally divided into two parts to handle the soluble nutrient in the runoff and in the sediment separately (21). For the soluble part, the general assumption is that the rate of change in concentration of soluble nutrients at the surface is proportional to the difference between existing concentrations and concentration in rainfall. The available nutrient content at the surface is a result of combining the residual nutrient at the surface with the amount from fertilizer application. The available nutrient due to rainfall is estimated from the nutrient concentration in rainfall and given as input data. The movement rates are evaluated using nutrient leaching and runoff extraction coefficients given as input data for the model.

The mathematical equations are used separately for each grid or cell where the total runoff, sediment, and nutrient quantities are calculated by adding the surface runoff contributions from various grids or cells to the base flow. The routing through the channel system is achieved using a storage-routing technique based on continuity. Deposition of sediment and decay of nutrients are estimated using fractions of the transported mass. Mixing cell models, like other transport models, is subject to numerical dispersion affected in part by cell size and the assumption of complete mixing. Appendix 2 summarizes some of the commonly used equations.

ASSESSMENT OF NUTRIENTS FROM A TYPICAL INDIAN RIVER BASIN

To calibrate and test the mathematical equation shown in Appendixes 1 and 2, the nonpoint-source pollutants from three agricultural watersheds, ranging from 2–5 km², of Kali Basin, western Uttar Pradesh, India have been analyzed. River Kali in western Uttar Pradesh, India, has a significant socioeconomic value for nearby areas. It receives many point and nonpoint sources of pollution (26–29). The study area (Fig. 2) is part of the Yamuna basin in the Indo-gangetic Plains, composed of Pleistocene and subrecent alluvium, and lies between 29° 13' to 30°N latitude and 77° 35' to 77° 45' E longitude. The mean rainfall across the basin is 1000 mm, which occurs mainly during the monsoon period. The basin area lies between elevations from 276 m above mean sea level to 221 m above mean sea level, and the major land use is agriculture. The soils of the area are loam to silty loam and are normally free from carbonates. Agricultural waste is also an important factor contributing to the pollution of the river water.

The basin boundary, drainage pattern, contour maps, spot heights and built-up area maps for River Kali, were digitized using ILWIS—GIS. For the analysis, the spatial data set available in grids or cells has been transformed

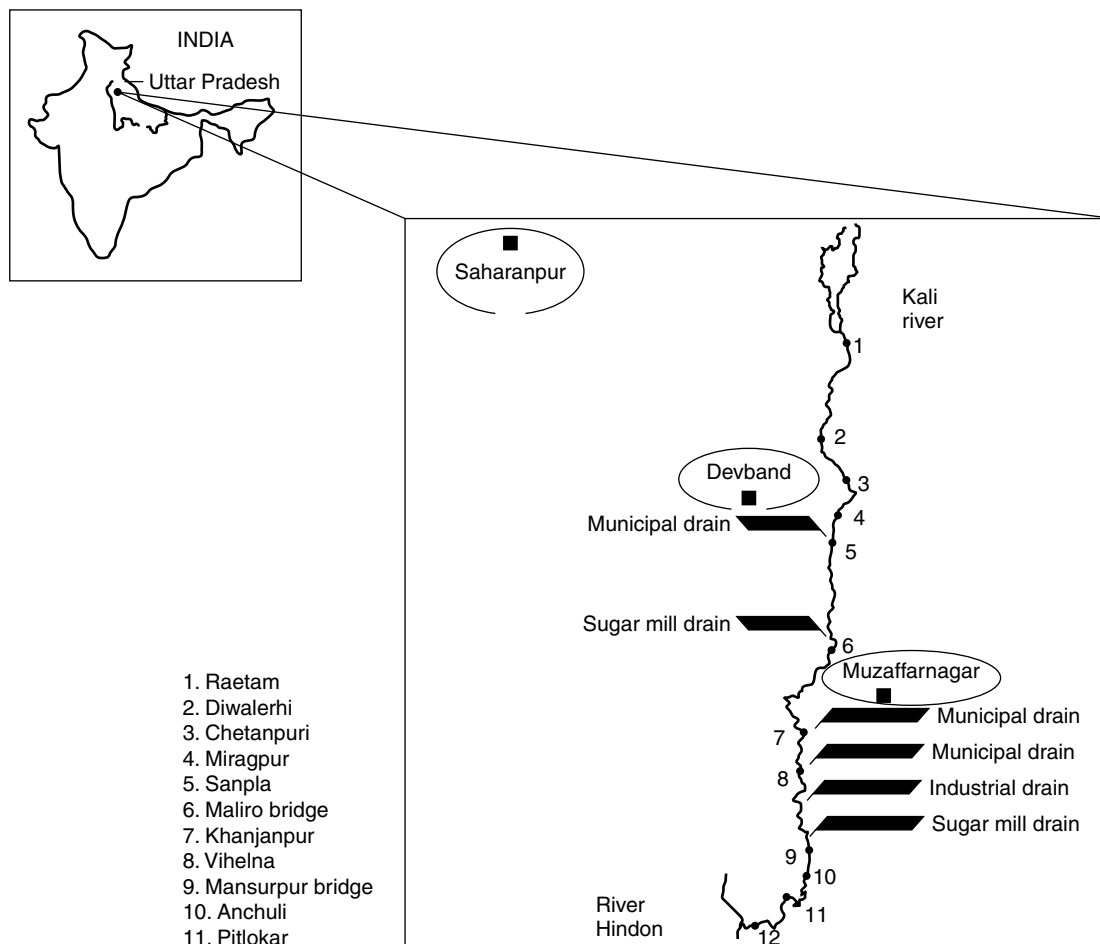


Figure 2. Location of River Kali in India.

to a digital elevation model (DEM), followed by filtering to arrive at a slope map, or a flow path map. For delineating land use/land cover maps, classification and quantification of different crops, and its temporal variation, IRS LISS III multispectral imageries and IRS-PAN data for the study period have been used. Ground truth verification was done to verify the results obtained using remote sensing data.

Further, extensive sampling from different sampling points giving a total of 576 data sets for nitrate (NO_3) and orthophosphate (o-PO_4) was done from March 1999 to February 2000. The nutrients were analyzed following a standard method (30). The NPS nutrient (NO_3 and o-PO_4) values obtained using mathematical equation based on the mixing cell concept, given in Appendixes 1 and 2, were compared with the observed nutrient values for all four agricultural watersheds (Fig. 3). It was found that the coefficient of determination, r^2 , between observed and computed values of nutrients was greater than 0.85 in all four agricultural watersheds. It was also found that the error estimate viz., standard error, was less than 2 for all cases.

SUMMARY

To estimate pollution from a large agricultural area, distributed parameter and continuous simulation models

are preferred because they simulate pollution producing events, total maximum daily pollution load, its seasonal variations, and represent the pollutant losses at different locations within the agricultural area. Remote sensing and GIS are important tools coupled with the mathematical models to estimate pollution from a large agricultural area. An illustrative case study from India demonstrates that NPS pollution can be computed well using the mixing cell concept with remote sensing and GIS support.

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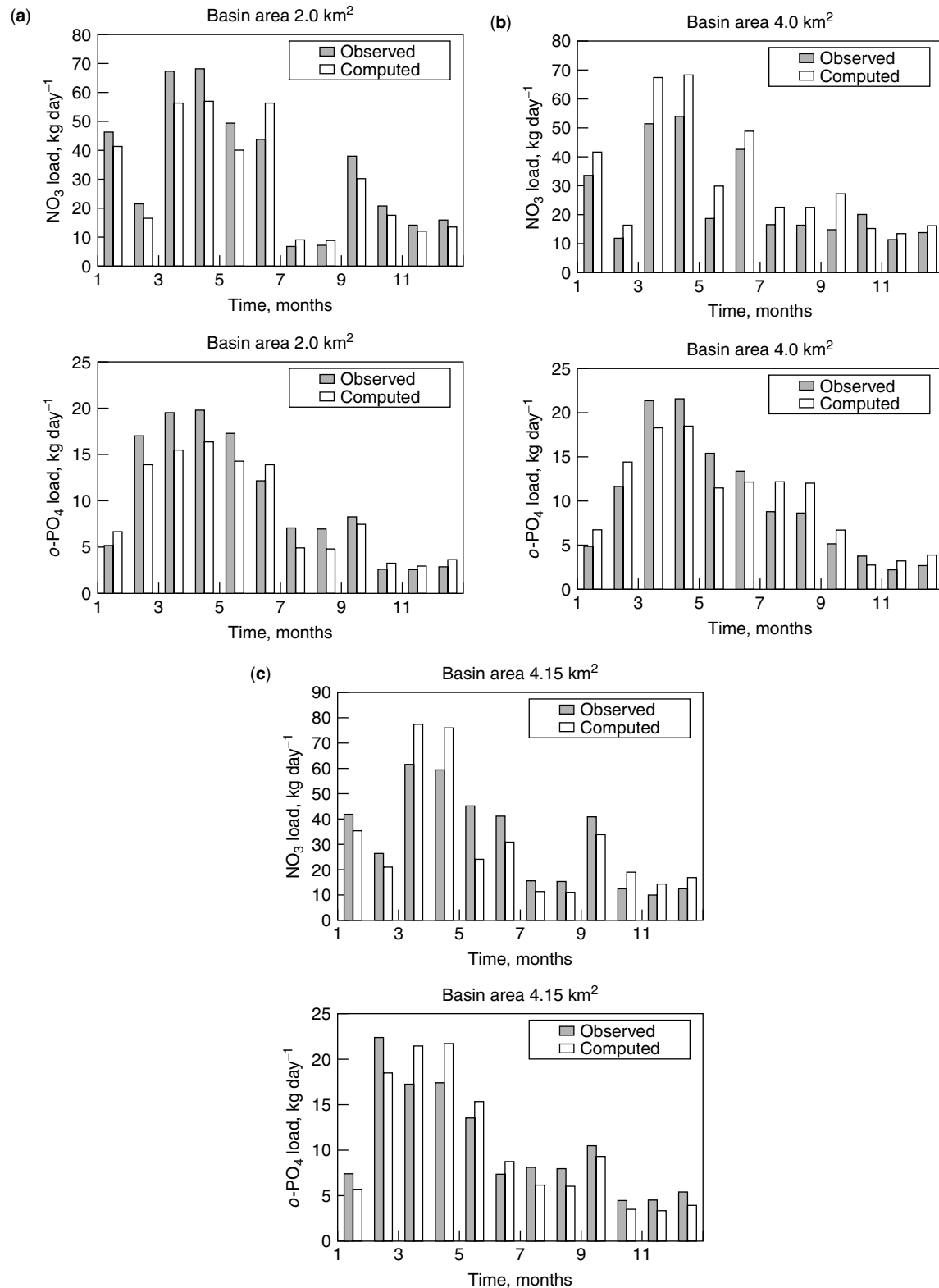


Figure 3. Observed and computed NPS nutrients from River Kali, India.

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APPENDIX 1

Specific Equations for Different Variables Used in NPS Estimation

Variables	Author(s)	Governing Equation ^a
Overland flow	Manning's formula	$Q_r = \frac{1}{R_3} (D_1 - D_s)^{5/3} S_1^{1/2} A$
Interflow	(31)	$Q_{\text{int}} = R_{\text{ec}} (W_{\text{ac}} - R_{\text{et}})$
Infiltration and surface detention	(32)	$\frac{dF}{dt} = k \left[1 + \left(\frac{(m - m_0)(P_{\text{ot}} + D_1)}{F} \right) \right]$
	(33)	$P_{\text{et}} = \alpha \frac{s(T_a)}{s(T_a) + \zeta} (K_n + L_n) \frac{1}{\rho \lambda_y}$
Interception	(34)	$V = (S_i + C_p E_a t_R) (1 - e^{-kP})$
Surface storage	(31)	$D_s = S_d (1 - e^{-kP_e})$
Evapotranspiration	(35)	$P_{\text{et}} = 0.0075 R_a C_{\delta_t}^{1/2} T_{\text{avg}}$
Transport capacity	(36)	$Y_C = 2.65 p c r_f$
Shear-stress relationships	(36)	$\tau_d = \frac{\beta}{\beta + 1} \left(\frac{60}{K_f} \right) \gamma H_L S_o$ $\tau_c = (\sigma - 1) \gamma \phi D_{50}$
Rainfall soil detachment	(36)	$G_{\text{rf}} = E_{\text{rf}} (1 - GC) CFD$
Runoff soil detachment	(36)	$G_{\text{ro}} = E_{\text{ro}} D$
Potential sediment supply	(36)	$Y_S = (G_{\text{rf}} + G_{\text{ro}}) \Delta t$
Soluble nutrient (nitrogen)	(21)	$C_{\text{RON}} = \frac{(N_{\text{AVS}} - N_{\text{AVR}})}{F_{\text{POR}}} * [e^{(-N_{\text{DMV}} I_{\text{EFF}})} - e^{(-N_{\text{DMV}} I_{\text{EFF}} - N_{\text{RMV}} R_{\text{OFF}})}]$ $+ \frac{N_{\text{RNC}} R_{\text{OFF}}}{P_{\text{EFF}}}$
Sediment attached nutrients	(21)	$N_{\text{SED}} = N_{\text{SCN}} Y_{\text{SED}} ER$ $P_{\text{SED}} = P_{\text{SCN}} Y_{\text{SED}} ER$ $ER = a Y_{\text{SED}}^b T_f$

V = interception depth (mm), S_i = storage capacity (mm), C_p = ratio of vegetated surface area, E_a = evaporation rate (mm h⁻¹), t_R = duration of the rainfall (h), k = constant (mm⁻¹), P = precipitation (mm), D_s = depression storage (mm), S_d = surface retention value (mm), P_e = cumulative rainfall excess (mm), F = total depth of infiltrated water (mm), t = time (s), K = saturated conductivity (mm s⁻¹), m = average moisture content of the soil, m_0 = initial soil moisture content, P_{ot} = capillary potential (mm), D_1 = detention storage (mm), P_{et} = potential evapotranspiration rate (mm d⁻¹), α = equilibrium factor, $s(T_a)$ = slope of the saturation vapor pressure vs. temperature curve, ξ = psychrometric constant, K_n = short-wave radiation, L_n = long-wave radiation, ρ = mass density of water, λ_y = latent heat of vaporization, R_a = total incoming solar radiation (mm), C_{δ_t} = temperature reduction coefficient, δ_t = temperature difference (°F ← °C), T_{avg} = mean temperature (°F ← °C), Q_{int} = interflow (m³ s⁻¹), R_{ec} = coefficient representing depletion, W_{ac} = water accumulation in the upper zone storage (mm), R_{et} = retained storage (mm), Q_r = channel inflow (m³ s⁻¹), $D_1 - D_s$ = runoff depth above ponding, R_3 = combined roughness and channel-length parameter optimized for each land class, S_1 = average overland slope, A = area of the element (m²), Y_C = sediment transport capacity (kg m⁻²), c = volumetric sediment concentration ratio = $A \left(\frac{\tau_d}{\tau_c} \right)^B$, A = 0.00066, B = 1.61, r_f = runoff (mm), τ_d = dominant flow shear stress, τ_c = critical stress, β = discharge parameter (= 5/3), γ = water specific weight (kg m⁻² s²), H_L = average runoff depth (mm), S_o = average overland slope, K_f = overland flow friction = 60 + 3140 GC^{1.65}, GC = ground cover factor, D_{50} = median size of soil particles (mm), σ = specific weight of sediment, ϕ Shields entrainment function = $\frac{0.11}{R^*} + 0.0211 \log_{10} R^*$, R^* = Reynolds number = $\frac{\sqrt{\tau_d} \gamma D_{50}}{\nu}$, ν = kinematic viscosity of water (m³ s⁻¹), G_{rf} = rate of soil detachment due to rainfall (kg m⁻² h), CF = canopy factor, D = soil erodibility factor (gJ⁻¹), E_{rf} = rate of rainfall energy (Jm⁻² h) = $i(11.9 + 8.7 \log_{10} i)$, i = rainfall intensity (mm h⁻¹), G_{ro} = rate of soil detachment due to runoff (kg m⁻² h), E_{ro} = rate of energy input to the soil by the flow (Jm⁻² h) = $\left(\frac{60}{K_f} \right) \gamma \frac{Q_L}{2}$, Q_L = unit flow discharge (m² h⁻¹), S_o = element slope, Δt = time increment (h), C_{RON} = soluble nitrogen concentration in runoff (kg ha⁻¹), N_{AVS} = available nitrogen content in the surface (kg ha⁻¹), N_{AVR} = available nitrogen in rainfall (kg ha⁻¹), N_{DMV} = rate of downward movement of nitrogen into soil, N_{RMV} = rate of nitrogen movement into runoff, I_{EFF} = effective infiltration (mm), R_{OFF} = total runoff (mm), F_{POR} = porosity factor, N_{RNC} = nitrogen contribution due to rain (kg ha⁻¹), P_{EFF} = effective precipitation (mm), N_{SED} = overland nitrogen transported by the sediment (kg ha⁻¹), N_{SCN} = soil nitrogen concentration (0.001 gNg⁻¹soil), ER = nutrient enrichment ratio ($a = 7.4$, $b = -0.2$), T_f = correction factor for soil texture (0.85 for sand, 1.0 for silt, 1.15 for clay, and 1.50 for peat).

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APPENDIX 2

Specific Equations for Routing Component of NPS Assessment

Variables	Author(s)	Governing Equation ^a
Grid or cell for sediment	(25)	$Y_{SEDout} = 1000[(1 - S_{Dep})(Y_{SEDab} + Y_{SEDin})]$
Grid or cell for nutrients	(25)	$CC_{RONout} = \frac{100}{R_{OFF}}[(1 - N_{Dec})(C_{RONab} + C_{RONin})]$

^aThe subscripts 1 and 2 indicate the beginning and end of the time step; $I_{1,2}$ = inflow to the reach ($m^3 s^{-1}$) consisting of overland flow, interflow base flow and channel flow from all contributing upstream basin elements; $O_{1,2}$ = outflow from the reach ($m^3 s^{-1}$); $S_{1,2}$ = storage in the reach (m^3); Δt = time step of the routing (s); R_2 = channel roughness parameter; A_x = channel cross-section area (m^2); S_o = channel slope; Y_{SEDout} = sediment leaving the cell (ppm); S_{Dep} = deposition fraction; Y_{SEDab} = sum of all the sediment entering the cell ($kg ha^{-1}$); Y_{SEDin} = sediment generated within the element; CC_{RONout} = soluble nutrients concentration in runoff leaving the cell (ppm); N_{Dec} = nutrients decay fraction; CC_{RONab} = sum of all the nutrient entering the cell ($kg ha^{-1}$); CC_{RONin} = nutrient generated within the element.

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DEEP-WELL TURBINE PUMPS

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Deep-well turbine pumps are used to pump groundwater to the surface, and in some cases, they may be used to pump water out of sumps and supply pressure. Deep-well turbines consist of a housing or bowl, impellers, and a shaft, all of which are installed in the well [Figs. 1(a) and 1(b)]. The housing or bowl contains both the impeller and the diffuser vanes. The shaft connects the impeller to the electric motor or engine, whereas the impeller transfers the energy developed by the motor or engine to the water. Diffuser vanes guide the flow of water from the impeller to the discharge point of the bowl.

Impellers transfer energy from the motor or engine to the water either by centrifugal force or by lifting action. They may be enclosed or semi-open. Enclosed impellers have a disk or shroud on both sides of the vanes (Fig. 2). These impellers require a close clearance between a wearing ring, located at the inlet of the impeller, and the impeller skirt of the bottom extension of the impeller. In semi-open impellers, the shroud is only at the top of the impeller. Semi-open impellers have a shroud only on the top (Fig. 2). A close clearance (0.076 mm–0.177 mm) between the bottom of the vanes and the pump housing is necessary for good performance. Semi-open impellers are less susceptible to wear from sand because they can be raised slightly to reduce abrasion. This process, however, will decrease pump output and efficiency.

Mixed flow impellers are often used in deep-well turbine pumps. Mixed flow impellers use both centrifugal and lifting forces to pump water. When the impeller rotates, centrifugal forces develop in the water inside the impeller and cause water to flow toward the impeller's outer edge. This flow, in turn, results in water flowing into the center or eye of the impeller. Rotating a mixed flow impeller also transfers energy to the water by a lifting force. Impellers that pump water only through centrifugal force are classified as radial flow impellers. Impellers that pump water only through lifting forces are classified as axial or propeller impellers.

Some deep-well turbines use a line-shaft to connect the impellers to an electric motor or engine. Line shafts are supported by bearings—either oil-lubricated or water-lubricated—spaced at intervals along the shaft length.

Oil-lubricated bearings are housed within a tubing inside the column pipe (the pipe that carries the water to the surface). An oil reservoir located on the pump head drips oil down the shaft.

Submersible pumps are deep-well turbines with a waterproof electric motor installed in the well below the pumping level. The motor is attached directly beneath the bowl assembly, and water intake is between the motor and the pump. A specially designed motor, slimmer and more compact than a line-shaft motor, is installed in the well. As both motor and pump are situated inside the well, submersible pumps do not require a line shaft.

In submersible pumps, it is crucial that enough water circulate around the motor to keep it cool. As water in the well should flow past the motor casing into the pump intake, submersible motors should be located above the level at which most of the flow enters the well. A shroud can be installed over the pump intake to force water to flow past the motor.

As the impellers and bowls must be submerged inside the well, the diameters of the impeller and bowl are limited by the diameter of the well, which in turn limits the output (capacity and pressure) developed by each impeller. This limitation can be overcome if the bowls are installed in stages, with each stage adding to the output developed by the preceding stages. A five-stage pump will, therefore, have five bowl assemblies, each containing an impeller, with the water guided from stage to stage by diffuser vanes built into the bowl assembly. Figure 1 shows a two-stage pump.

Because deep-well turbine pumps are installed in the well, pump size is limited by the diameter of the well casing. If the pump is too large for the casing, rocks from the gravel pack can become lodged between the casing, which makes it difficult to remove the pump and—particularly if the well is crooked—the pump may be difficult to install. The clearance between the bowls and the well casing should be at least 25 mm. Narrower clearances may complicate installation or, in crooked wells or where gravel becomes lodged between the pump and casing, may make it difficult for the bowls to be removed for repair.

The output of the pump must be sufficient to lift the water to the ground surface and provide the desired pressure at the desired flow rate or pump capacity. However, in some cases, deep-well turbines lift the water only and a centrifugal pump located at the discharge point of the well, commonly called a booster pump, is placed in line with the deep-well turbine pump to supply pressure.

The output of a pump is described by its performance characteristic curves, developed by pump manufacturers. These curves show the amount of total dynamic head developed by a single stage at a particular capacity, the pump efficiency at that capacity, and the horsepower requirements of the pump. Performance curves show which pump will provide the desired total head and capacity and determine what size electric motor or engine the pump will require.

Figure 3(a) shows the total head and capacity performance curve for a typical pump. In a no-flow condition (shut-off), the total head is maximum, or 22 m for this

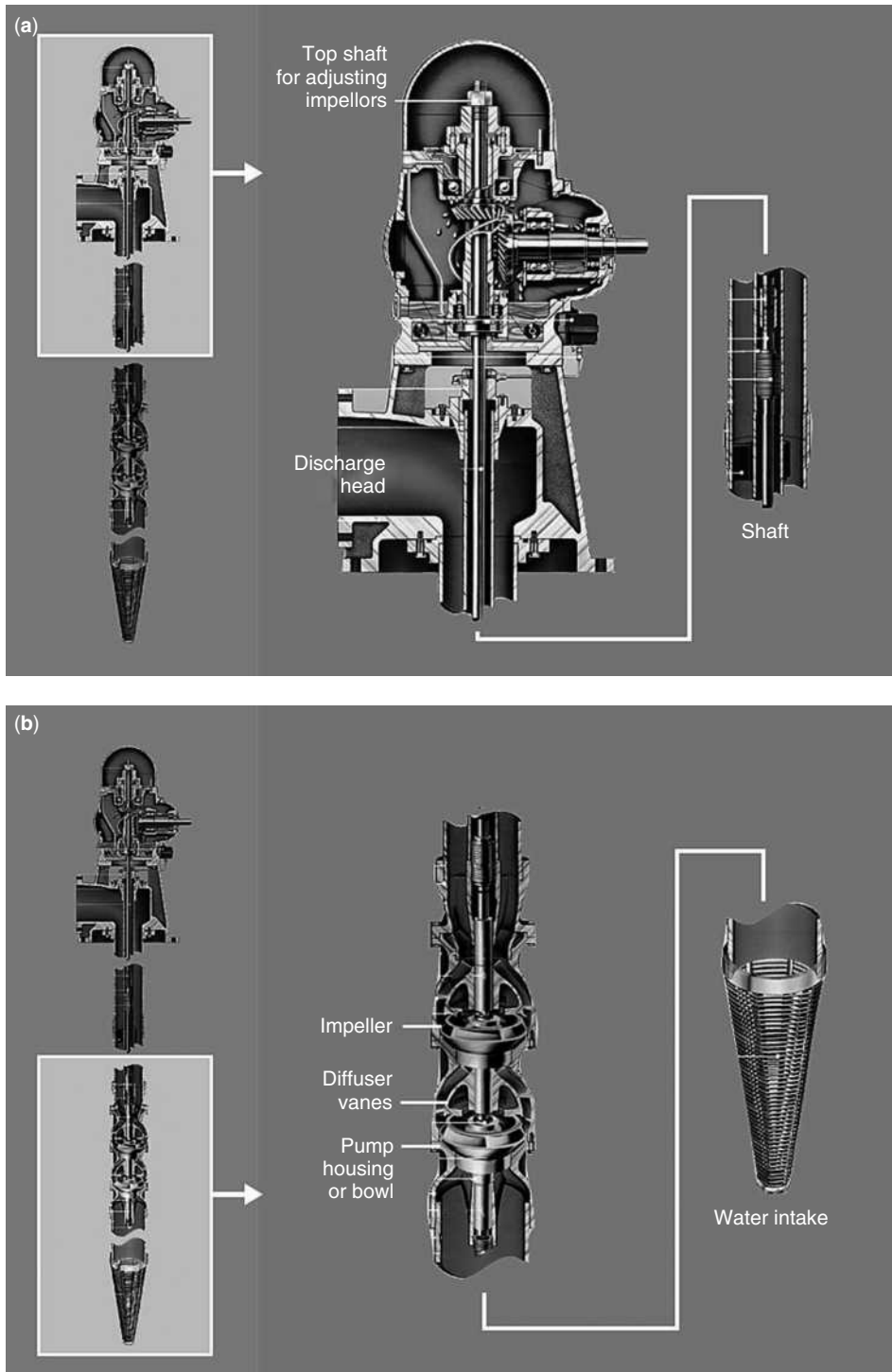


Figure 1. Cutaway of a deep-well turbine pump that shows (a) the pump head, gear head, and shaft; and (b) pump housing, impellers, and shaft. [Source: Hanson (1)].

pump. As the pump capacity increases, the total head decreases. At low capacities, most of the pump output consists of total head (normally measured as the sum of the pumping lift and the discharge pressure head), whereas at high capacities, most of the pump output is in capacity.

Pump efficiency increases as pump capacity increases until a maximum efficiency is reached [Fig. 3(b)]. Thereafter, pump efficiency decreases as capacity continues to increase.

Metric horsepower (mhp) increases slightly as capacity increases to a maximum, and then it may decrease slightly [Fig. 3(c)].

The performance curves are only for one-stage, but they can develop performance curves for multiple-stage pumps. For example, Fig. 3(a) shows that in a one-stage pump, 14.6 m of head is developed at 3028 L/min. The mhp is 14.2. In a three-stage pump, the head developed at 3,028 L/min will be three times that of a single stage,

or 43.8 m. The mhp will also be tripled, to 42.6 mhp, and the pump efficiency may change slightly. Usually the manufacturer's curve will show the efficiency adjustments. The performance curve for a multiple-stage pump can therefore be estimated by multiplying the head and bhp of the performance curve by the number of stages for each capacity.

Pump performance curves have a variety of shapes. Some head/capacity curves are steep (large decreases in head occur as capacity increases), whereas others are flat (small decreases in head occur as capacity increases). The best curve will depend on changes in pumping conditions. Where pumping levels fluctuate significantly, a steep curve may allow for significant changes in pumping lift with only slight changes in pump capacity. A flat curve may help to maintain a relatively constant pressure over a wide range of flow rates.

An initially efficient pump with the necessary head and capacity can become inefficient. Inefficiency is frequently caused by wear because of sand in the well water, but it can also be caused by a mismatch between the pump and water system characteristics because of long-term changes in groundwater levels or changes to the water supply system. A mismatched pump is one that is operating properly but at an efficiency much less than the potential maximum efficiency. "Repairing" a mismatched pump will not improve efficiency.

The pump's performance can be evaluated by first testing the pump and then comparing that information with the manufacturer's performance curves. The pump test consists of measuring the pumping lift, discharge pressure, capacity, and input power to the pump and then calculating overall pumping plant efficiency—that is, the combined efficiency of the pump and motor or engine. The overall efficiency will be less than the pump efficiency reflected in the manufacturer's performance curve because of the inefficiency of the motor or engine.

Pump tests are conducted by utility companies, pump dealers, and consultants. In defining the current status of the pump, the test may provide any or all of the following information:

- Pumping lift or level—elevation difference between the discharge pipe at the pump head and the water level in the pumping well

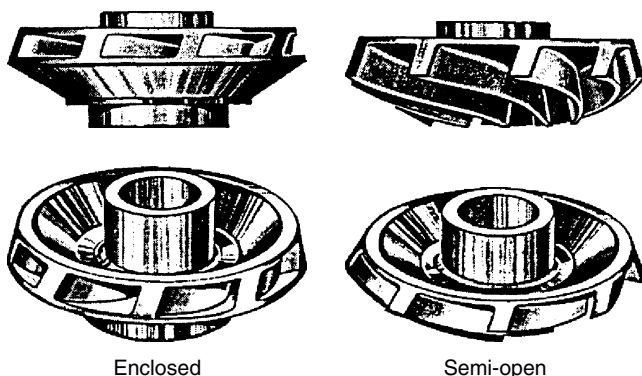


Figure 2. Enclosed and semi-open impellers. [Source: Hanson (2)].

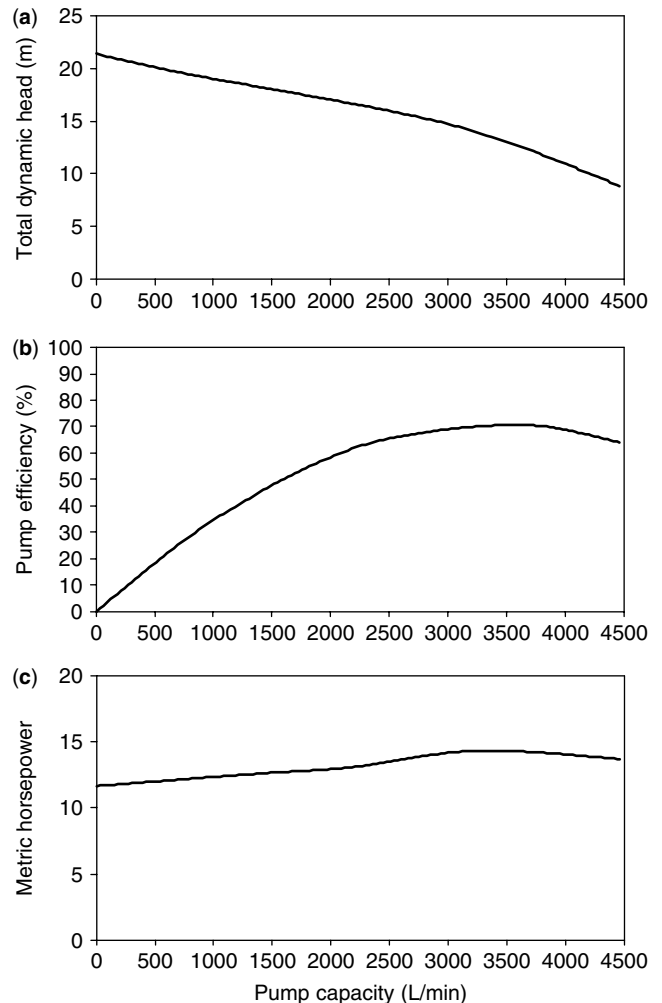


Figure 3. Pump performance curves that show the relationships between (a) total head and capacity, (b) pump efficiency and capacity, and (c) metric horsepower and capacity.

- Static or standing water level—elevation difference between discharge pipe and water level in the well before pumping
- Drawdown—difference between the pumping level and the static water level
- Discharge pressure—pressure at the discharge point of the pumping plant measured with a pressure gauge
- Discharge pressure head—the height of the column of water needed to provide the pressure at its base—calculated by multiplying the pressure by a conversion constant (0.704, 0.1021, or 10.21 for discharge pressure in psi, kPa, or bars, respectively)
- Total head—sum of the pumping lift and the discharge pressure head (note that if the pump is situated deep in the well, pressure losses caused by friction in the column pipe may have to be added to the total head)
- Pump capacity—water flow rate of the pumping plant.

- Well yield—flow rate or capacity divided by the drawdown (a well yield decreasing over time may indicate plugging in the well)
- Input horsepower to electric motor
- Percent motor overload
- Input kilowatts to motor
- Kilowatt-hours per acre-foot of pumped water
- Overall pumping plant efficiency

The pumping lift is measured by lowering an electric water level indicator (two electrodes attached to a wire connected to an ampere meter) or some other measuring device into the space between the well casing and the column pipe. In some cases, an air line is permanently installed to measure the pumping lift. This measurement requires access to the inside of the well casing; without such access, the pump test cannot be performed.

The pump capacity is measured with a flow meter installed in the discharge pipe. To operate properly, a flow meter requires a straight section of pipe eight to ten pipe diameters long immediately upstream from the meter to prevent or reduce turbulence in the water. Too much turbulence from valves, bends, or elbows will prevent accurate flow measurements. It is also recommended that a straight section of pipe two pipe diameters long be installed immediately downstream from the flow meter.

The input power of an electric motor or engine must be determined to calculate the overall pumping plant efficiency. Input horsepower of electric motors can be easily calculated. In some cases, the power meter will provide this information. Determining the input horsepower of engines is much more complicated and will require measuring the fuel flow rate to the engine.

The overall pumping plant efficiency can be calculated by the following equation:

$$E_o = (Q \times H) / (4634 \times \text{mhp})$$

where E_o is the overall pumping plant efficiency, Q is the capacity (L/min), H is the total head (m), and mhp is the metric input horsepower.

The pump test data should be compared with the manufacturer's performance curves to determine how the pump's actual performance compares with the manufacturer's recommended performance. This comparison step is important because it is possible to have a pumping plant measuring low efficiency but nonetheless operating properly. Repairing such a pump would be of no benefit. If the operating point is close to the head/capacity performance curve, the pump is operating properly, regardless of the overall efficiency calculated from the test data. If a significant difference exists between the performance curve and the operating point, the pump may be worn and should be repaired or replaced.

The pump test data shown in Fig. 4 illustrate the evaluation process. Two pump tests were conducted—the first when the pump was worn, with an overall efficiency of 39% and the second after the pump was repaired, with an overall efficiency of 50%. The worn pump showed a significant difference between the manufacturer's curve

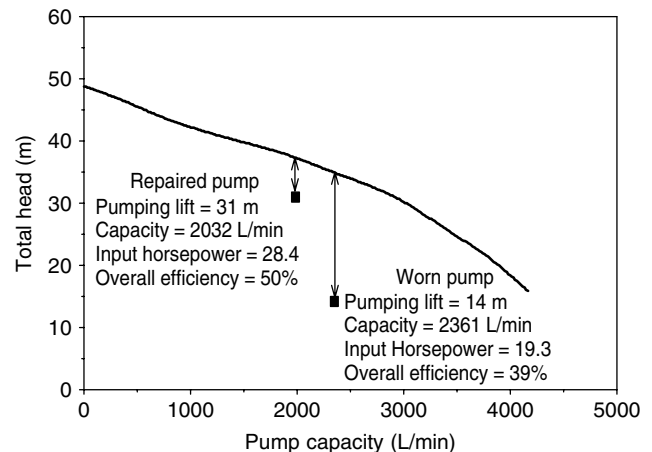


Figure 4. Comparison of pump test data with manufacturer's total head-capacity curve.

and the operating point, whereas the repaired pump showed only a slight difference. However, the efficiency of the repaired pump suggests that the pump is mismatched to its operating conditions and probably should be replaced by a matched pump.

Adjusting the impellers can partially restore the capacity and total head of deep-well turbine pumps using semi-open impellers that are slightly or moderately worn. The adjustment consists of slightly lowering the pump shaft to which the impellers are connected by turning the nut at the top of the motor or gear head. The adjustment must also account for line shaft elongation that occurs when the pump is operating.

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MICROBIAL QUALITY OF RECLAIMED IRRIGATION: INTERNATIONAL PERSPECTIVE

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Wastewater reuse has been practiced in different forms throughout the history of human civilization. Previously, sewage had been dumped in rivers and lakes or used locally to irrigate crops. Explosive population growth and diminishing fresh water resources around the globe have now made wastewater a valuable resource, especially in arid and semiarid areas of the world. The introduction of modern wastewater treatment processes has given new dimensions to water reuse. Reclaimed waters are commonly used for landscape or recreational irrigation (parks, golf courses, athletic fields, school yards, highway

medians, and public and private lands) and agricultural crop irrigation (field crops, vegetables, orchids, and tree plantations).

Human feces, which may have microbial constituents as high as 10–30%, are one of the major sources of infectious and noninfectious agents found in municipal wastewater. Thus, wastewater can contain high concentrations of excreta-related microorganisms. The principal microorganisms (infectious agents) that may be present in raw sewage can be classified into three broad groups: bacteria, parasites (protozoa and helminths), and viruses. During an outbreak, pathogen numbers in local sewage go up. More than 30 excreta-related diseases of public health importance have been documented, and many of these are of specific importance in wastewater reuse schemes. Table 1 summarizes the major infectious agents potentially present in raw domestic wastewater and their respective diseases.

TYPES OF MICROORGANISMS

Bacteria

Bacteria are microscopic organisms, ranging from approximately 1 to 6 μm in length. Human and animal intestinal tracts are colonized by different types of bacteria, which are routinely shed in the feces. The number of bacteria in feces can be in the range of 10^8 – 10^{11} per gram of fecal material, and these bacteria end up in municipal

wastewater (1). Coliforms, such as *Escherichia coli*, and *Enterobacter* constitute a small group of intestinal bacteria, which has historically been used as an indication that an environment is contaminated by human excreta. In addition to coliforms, some pathogenic bacteria, such as *Salmonella*, *Shigella*, *Vibrio*, *Mycobacterium*, *Clostridium*, *Leptospira*, and *Yersinia*, can also colonize part of a gastrointestinal tract, resulting in gastroenteritis.

A number of opportunistic bacterial pathogens can also be found in waste and reclaimed water. These opportunistic pathogens include *Pseudomonas*, *Streptococcus*, *Flavobacterium*, and *Aeromonas* species (2).

Parasites

Parasites are a very diverse group of microorganisms; they can be further categorized as protozoan, metazoan, or helminth. Parasites often have a complex life cycle sometime requiring a stage in intermediate hosts. The environmental stage of parasites found in wastewater is normally called cyst/oocyst (protozoans) or egg (helminth). In general, parasite cysts or eggs are larger than bacteria. Parasites are present in the feces of infected persons; however, healthy carriers may also excrete them. Cysts do not reproduce in the environment but can survive in soil for months or even years, depending on environmental conditions.

Entamoeba histolytica, the most dangerous protozoan parasite, occurs worldwide, although the incidence in the

Table 1. Pathogenic Microorganisms Associated with Wastewater

Group of microorganisms	Examples	Disease
Bacteria	<i>Campylobacter jejuni</i>	Gastroenteritis (campylobacteriosis)
	<i>Clostridium</i>	Gastroenteritis
	Enteropathogenic	Gastroenteritis
	<i>Escherichia coli</i>	
	<i>Leptospira</i>	
	<i>Mycobacterium</i>	
	<i>Salmonella typhi</i>	Typhoid fever
	<i>Salmonella</i> sp.	Gastroenteritis (salmonellosis)
	<i>Shigella</i>	Bacillary dysentery
	<i>Vibrio cholera</i>	Cholera
Viruses	<i>Yersinia</i> sp.	Gastroenteritis (yersiniosis)
	Adenoviruses	Gastroenteritis (conjunctivitis, Respiratory)
	Caliciviruses	Gastroenteritis
	Coxsackie viruses	Flu-like symptoms, gastroenteritis
	Echoviruses	Flu-like symptoms, aseptic meningitis
	Hepatitis A virus	Infectious hepatitis
	Hepatitis E virus	Infectious hepatitis
	Polioviruses	Poliomyelitis
	Reoviruses	Respiratory diseases
	Rotavirus	Acute gastroenteritis
Protozoan	<i>Balantidium</i>	Balantidiasis (gastroenteritis)
	<i>Cryptosporidium</i>	Cryptosporidiosis (diarrhea)
	<i>Entamoeba coli</i>	Diarrhea, ulceration
	<i>Entamoeba histolytica</i>	Amebiasis (amoebic dysentery)
	<i>Giardia lamblia</i>	Giardiasis (diarrhea)
Helminth	<i>Ascaris</i> sp.	Ascariasis (roundworm infection)
	<i>Necator americanus</i>	Ancylostomiasis (hookworm infection)
	<i>Taenia</i> sp.	Taeniasis (tapeworm infection)
	<i>Trichuris trichuria</i>	Trichuriasis (whipworm infection)

United States is not very well documented. *Cryptosporidium parvum* oocysts are the most important biological water contaminants in the United States (3).

Several helminthic parasites may be found in wastewater. *Ascaris lumbricoides* (roundworm) is one of the most important human parasites. It is estimated that approximately 25% of the world's population is infected by this nematode (4).

Viruses

Viruses are smaller than bacterial disease agents and range in size from approximately 20 to 200 nm. They are obligate intracellular parasites that can multiply only within a host cell. Viruses that can replicate in the gastrointestinal tract of humans or animals are referred to as enteric viruses. The presence of enteric viruses in sewage is a matter of concern because they can be transmitted through the fecal-oral route. More than 120 different human enteric viruses have been reported to date. An infected person can excrete as many as 10^6 – 10^{10} plaque-forming units (pfu) of enteric viruses per gram of fecal material.

The most important human enteric viruses are enteroviruses (polio, echo, and Coxsackie), caliciviruses, rotaviruses, reoviruses, parvoviruses, and adenoviruses. Caliciviruses are the most frequently reported and documented that are transmitted by contaminated water. Even though the etiologic agent cannot be successfully cultivated in the laboratory, there is epidemiological evidence that caliciviruses are the most common cause of viral gastroenteritis. Hepatitis A, the virus that causes infectious hepatitis, is one of the most resistant to environmental stresses (5). Rotaviruses are the most infectious of all enteric viruses (6) and thus can be considered a high risk to health if present in wastewater.

MECHANISM OF DISEASE TRANSMISSION

There are several ways in which an individual can acquire disease from wastewater use, and those can be broadly categorized into two modes; direct ingestion and indirect ingestion. Humans can acquire an etiologic agent either directly by contact, ingestion, or inhalation of reclaimed irrigation water or indirectly by contact with the objects previously contaminated by the reclaimed water. Following are some of the conditions for a successful infection or disease from exposure to reclaimed wastewater.

- **Etiologic Agent:** The disease-causing agent (bacteria, virus or a parasite) must be present in wastewater and hence, in the wastewater from that community.
- **Survival of Etiologic Agent:** The disease-causing agent must survive all wastewater treatment processes to which it is exposed.
- **Exposure:** The disease competent person must come in contact, directly or indirectly, with reclaimed water carrying the disease agent.
- **Infectivity and Infective Dose:** The etiologic agent must be infectious, moreover, the disease-causing

agent must be sufficiently numerous at the time of contact to cause illness.

The exposure to an etiologic agent does not always result in disease. Disease results from a series of complex interactions between the host and the infectious agent, which can be defined in terms of the pathogenicity and virulence of the disease-causing agent and the relative susceptibility of the host. The susceptibility of the host depends on the general health of the subject and the specific pathogen. Age and the nutritional and immunological status of subjects are some of the other factors that determine the outcome of an infection.

MONITORING MICROORGANISMS

The detection, isolation, and identification of all microorganisms in a wastewater sample are very difficult, time-consuming, and cumbersome. The detection efficacy and sensitivity are influenced by various physicochemical parameters of the water sample, which further undermine the significance of such a strategy. To avoid the necessity of undertaking such a huge task, which has no equivalent value, indicator microorganisms are used to estimate the presence of fecal pathogens in water samples. An indicator should be present in equal or higher numbers and be more resistant than pathogens. Historically, fecal coliforms, in particular, *E. coli*, have been used as indicators of fecal contamination (7).

REMOVAL OF MICROORGANISMS BY WASTEWATER TREATMENT PROCESSES

The purpose of most advanced treatment processes is to remove either inorganic or organic constituents. Therefore, the removal of biological contaminants by these processes is only incidental in many cases and, generally is not too great. Microbial removal efficiencies of some treatment processes are given Table 2. Conventional biological treatment processes (trickling filters, activated sludge, and oxidation ponds) reduce the quantities of biological organisms in raw or settled sewage but do not eliminate them.

The most important treatment process for pathogen destruction is disinfection. In the United States, the most common disinfectant for both water and wastewater is chlorine. The efficiency of disinfection with chlorine depends on the water temperature, pH, length of contact, degree of mixing, presence of interfering substances, concentration and form of the chlorinated species, and nature and concentration of the organisms to be destroyed.

TYPES OF RECLAIMED IRRIGATION PRACTICES

Reclaimed irrigation can be practiced in the following five different ways:

- **Flood irrigation,** which wets almost all the land surface;

Table 2. Concentration of Various Microorganisms in Raw and Treated Wastewater^a

Microorganisms	Number per 100 Raw Sewage	Reduction in Number of Microbes (log ₁₀)		
		Primary Effluent	Secondary Effluent	Tertiary Effluent
Fecal coliform (MPN)	10 ⁹	2	1	6
<i>Salmonella</i> (MPN)	8 × 10 ³	1	2	1
Enteric viruses (PFU)	5 × 10 ⁴	0.5	1	5
Helminth ova	8 × 10 ²	1	2	1
<i>Giardia</i> cyst	10 ⁴	0.5	0.5	3

^aReference 8.

- Furrow irrigation, which wets only part of the ground surface;
- Sprinkler irrigation, in which the soil is wetted in much the same way as by rain;
- Surface drip irrigation, in which water is applied at each individual plant at an adjustable rate;
- Subsurface drip irrigation, in which the topsoil remains relatively dry but the subsoil is saturated.

CONTROL OF HUMAN EXPOSURE

Four groups of people are at potential risk from reclaimed irrigation:

- agricultural field workers and their families;
- crop handlers;
- consumers (of crops, meat, and milk);
- population living near the affected fields; and
- recreational area visitors, players, etc.

GUIDELINES AND TREATMENT PRACTICES FOR WASTEWATER REUSE IN AGRICULTURE

All reuse guidelines or criteria have a single basic objective, to ensure the safety of public health. Wastewater is known to harbor a variety of different microorganisms, influenced by various environmental factors, so it is possible to address different aspects of this issue. Reuse guidelines can be based on the level of indicator microorganisms, epidemiological and/or risk models, and treatment practices.

Early guidelines for wastewater reuse in agriculture were concerned only with the pathogen content of wastewater. A World Health Organization (WHO) meeting of experts on the reuse of effluents in 1971 reviewed the information concerning health risks and concluded that only a limited health risk would result from unrestricted crop irrigation with wastewater containing no more than 100 coliforms per 100 mL. Following the recommendation of the WHO meeting in 1971, new epidemiological evidence was accumulated, and earlier studies were reevaluated. A meeting in Engelberg in 1985 of public health experts, environmental scientists, and epidemiologists reviewed the work that had been done and concluded that the risk of infection was lower than previously thought, particularly, from bacterial and viral pathogens (9).

The WHO guidelines for wastewater reuse, which were proposed by the Engelberg meeting in 1985 and reaffirmed in 1989, were aimed at providing a high degree of health safety to workers and consumers, but at the same time to be realistic in the light of resource constraints and limited technology in many countries.

In the United States the State of California has a long history of reclaimed water regulation. The relatively stringent California standards are not economically or technologically feasible for developing countries.

INTERNATIONAL STANDARDS

The WHO water reuse guidelines (Table 3) have served as a baseline for the development of reclaimed irrigation standards throughout the world generally and in developing countries particularly. On the other hand, California criteria, which represent the higher end, are extensively followed by developed countries. Different countries have adapted both standards to develop their regional microbial standards for reclaimed irrigation (Table 4).

Italy became the first European country to regulate reclaimed irrigation when it adopted wastewater reuse guidelines in 1977 under its 1976 national Water Law. Although the Italian guidelines were published 1 year earlier than California guidelines, both essentially follow the same approach. The region of Sicily has adopted different regulations, which are much closer to the WHO approach. As an added precaution, this regulation forbids irrigating vegetables that are eaten raw with any type of reclaimed wastewater.

Cyprus produces approximately 25 Mm³ of urban effluent every year. Government has planned to expand irrigated agriculture by introducing reclaimed irrigation. Provisional standards for reclaimed water irrigation were introduced to regulate reclaimed irrigation, which take into account the special ecological and environmental conditions of Cyprus. These standards, which are stricter than the WHO guidelines, ensure the best possible application of the effluent for irrigation.

In France, reclaimed irrigation of crops is practiced mostly around big cities as the easiest way to treat and dispose of wastewater. In the early 1990s, the growing demand for irrigation water along with periodic droughts resulted in increased interest in water reuse. French health authorities [the Conseil Supérieur diHygiène Publique de France (CSHPF)] issued the "health guidelines for reuse, after treatment, of wastewater for crop

Table 3. Microbiological Quality Guidelines for Wastewater Use in Agriculture^a by WHO

Category	Reuse Conditions	Exposed Group	Intestinal Nematodes (Mean Arithmetic Number of Viable Eggs per liter)	Faecal Coliforms (Geometric Mean, Number per 100 mL)	Wastewater Treatment Expected to Achieve the Required Microbiological Quality
A	Irrigation of crops likely to be eaten uncooked, sports fields, public parks	Workers, consumers, public	1	1000	A series of stabilization ponds designed to achieve the microbiological quality indicated, or equivalent treatment
B	Irrigation of cereal crops, industrial crops, fodder crops, pasture and trees	Workers	1	No standard recommended	Retention in stabilization ponds for 8–10 days or equivalent helminth and fecal coliform removal
C	Localized irrigation of crops in category B if exposure of workers and the public does not occur	None	Not applicable	Not applicable	Pretreatment as required by irrigation technology, but not less than primary sedimentation

^aReference 10.**Table 4. Guidelines and Criteria for Wastewater Reuse in Irrigation in Various Countries^a**

Country	Crop Type	Treatment Required	Microbial Criteria (max.)
Australia	Residential use, municipal irrigation, unrestricted crop irrigation	Secondary, tertiary, filtration and disinfection	<10 thermotolerant coliforms/100 mL
	Ornamental ponds with public access, restricted crop irrigation, irrigation of pasture and fodder crops for dairy animals, fire fighting	Secondary and disinfection	<100 thermotolerant coliforms/100 mL
	Municipal irrigation with restricted crop irrigation, irrigation of pasture and fodder crops for grazing animals	Primary sedimentation and lagooning or secondary plus disinfection (if required)	<1000 thermotolerant coliforms/100 mL
	Irrigation for turf production, silviculture, nonfood chain aquaculture	Primary sedimentation and lagooning or secondary	<10,000 thermotolerant coliforms/100 mL
Israel	Cotton, sugar beet, dry fodder seeds, forest irrigation	None	None
	Green fodder, nuts, citrus, olives, peanuts, bananas, almonds, etc.	None	None
	Deciduous fruits, conserved vegetables, cooked and peeled vegetables	Disinfection with chlorination (60 minutes contact time)	250 coliforms/100 mL
	Unrestricted crops, including vegetables eaten raw, parks, and lawns	Sand filtration and disinfection with chlorination (60 minutes contact time)	12 coliforms/100 mL (80%) 2.2 coliforms/100 mL (50%) 0.0 <i>E. coli</i> /100 mL
Japan	Landscape irrigation	Disinfection (>0.4 mg/L residual chlorine)	10,000 coliforms/100 mL
Kuwait	Fodder, food crops not eaten raw, forest land	Advanced water treatment (1 mg/L residual chlorine after 12 h@20 °C)	100 coliforms/100 mL
	Food crops eaten raw	Advanced water treatment (1 mg/L residual chlorine after 12 h@20 °C)	2.2 coliforms/100 mL
Saudi Arabia	All irrigation purposes (unrestricted)	Advanced water treatment	<1 intestinal nematode egg/100 mL
	All irrigation purposes	Treatment plants and waste settling ponds	<1000 fecal coliforms/100 mL
South Africa	Irrigation of dry fodder crops, seed crops, trees, non-recreational parks, nurseries (restricted access)	Primary and secondary	<1000 fecal coliforms/100 mL
	Food crops not eaten raw, cut flowers, orchards and vineyards, pastures, parks	Primary, secondary, and tertiary; oxidation pond system	0.0 fecal coliforms/100 mL
	Pasture for milking animals	Primary, secondary, and tertiary	Drinking water standards
	Food crops eaten raw, lawns, nurseries (unrestricted access)	Advanced (general) drinking water standards	

^aReference 11.

and green spaces irrigation”(12). The guidelines stipulate maintaining of extensive water quality data. Wastewater reuse authorizations are granted on a case-by-case basis after review of a highly detailed dossier of French reclaimed irrigation regulations based on WHO guidelines but complement them with strict rules of application such as a minimum distance required between irrigation sites and residential areas or roadways.

In Mexico, wastewater is extensively used to irrigate crops. Because a very significant part of Mexican farm produce is sold in U.S. markets, Mexico has developed very comprehensive reclaimed irrigation regulations that are stricter than WHO suggested standards (Tables 5 and 6).

Israel has developed very comprehensive reclaimed water usage regulations because reclaimed water constitutes a significant part of its national water budget. About 92% of the total wastewater is collected, and 72% is reclaimed for various purposes. Almost 42% of total wastewater is reused for irrigation (13). The Israel Ministry of Health has established criteria for reclaimed irrigation and water reuse that are stricter than WHO standards. According to Israeli standards, settled effluent can be used on industrial and fodder crops, pastures and hay, vegetables eaten cooked, fruit trees, ornamental plants, or seed plants. Irrigation should be stopped 1 month before the harvest of apples, pears, and plums; broccoli and cauliflower when furrows are irrigated, and tomatoes used for canning if furrows are irrigated.

WHO guidelines have been debated for their effectiveness to protect public health. A large part of the answer probably lies in the treatment requirements of limit values. In spite of their safety, the stringency of the California standards is acting as a barrier to their widespread adoption worldwide. The WHO point of view is often criticized as being too lax, but the California approach is too technologically intensive and expensive for developing countries. One must realize that where

Table 6. Crops Regulated by 1991 Mexican Reuse Guidelines

Salad Vegetables	Vegetables	Herbs	Fruits
Beet root	Beet	Coriander	Blackberries
Celery	Broccoli	Epazote	Melon
Cucumber	Cabbage	Mint	Strawberries
Lettuce	Carrot	Parsley	Sweet turnip
Radish	Cauliflower		Watermelon
Red tomato	Courgette		
Watercress	Garlic		
	Green tomato		
	Mushrooms		
	Onion		
	Papalo		
	Spinach		
	Wild greens		

raw wastewater is directly reused, a widespread practice in many developing countries, the WHO guidelines are already a major step forward.

However, other requirements also seem to be necessary to complement WHO guidelines. Based on an extensive analysis of existing guidelines worldwide, the need for developing health-related chemical criteria for land application of reclaimed wastewater has also been reported by the WHO (14). This, added to the fact that the controversy between the ‘WHO camp’ and the ‘California camp’ has now been raging for a number of years means that the time has come to update the existing guidelines on an international basis. The door is therefore now open worldwide for a ‘third approach’ integrating the epidemiological knowledge base generated since 1989 and the latest technological developments in wastewater treatment.

Table 5. Conditions for Irrigation of Restricted Crops Under 1991 Mexican Guidelines

Type of Irrigation	Type of Wastewater ^a	Minimum Time Between the Last Irrigation and the Harvest (days)	Crops/Banned Crops
Flood	1	20	All crops listed in Table 6, except garlic, cucumber, sweet turnip, melon, and watermelon
	2	20	All crops listed in Table 6, except melon and watermelon
	3	20	All crops listed in Table 6
	4	20	All salad crops, vegetables, and fruits in general
Furrow	1	15	All crops listed in Table 6, except garlic, cucumber, sweet turnip, melon, watermelon, and green tomato
		20	No restrictions
	2	20	All crops listed in Table 6, except garlic, cucumber, sweet turnip, melon, watermelon, and green tomato
	3	20	All crops listed in Table 6, except melon and watermelon
Spray	4	20	All salad crops, vegetables, and fruits in general
	1	20	All crops listed in Table 6, except garlic, cucumber, sweet turnip, melon and watermelon
	2,3,4	20	All salad crops, vegetables and fruits in general

^aWastewater Type 1: <1000 total coliforms/100 mL and 0 helminth eggs/liter;

Wastewater Type 2: 1–1000 fecal coliforms/100 mL and ≤1 helminth eggs/liter;

Wastewater Type 3: 1001–100,000 fecal coliforms/100 mL, and no restriction on helminth eggs;

Wastewater Type 4: >100,000 fecal coliforms/100 mL and no restriction on helminth eggs.

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SOIL SALINITY

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DEFINITION, SOURCES, EFFECTS, AND GLOBAL IMPACTS

Soil salinity refers to the presence of major dissolved inorganic solutes in the soil solution (i.e., aqueous liquid phase of the soil and its solutes), which consist of soluble and readily dissolvable salts, including charged species (e.g., Na^+ , K^+ , Mg^{2+} , Ca^{2+} , Cl^- , HCO_3^- , NO_3^- , SO_4^{2-} , and CO_3^{2-}), nonionic solutes, and ions that combine to form ion pairs. The primary source of salts in soil and water is the geochemical weathering of rocks from the earth's upper strata, with atmospheric deposition and anthropogenic activities serving as secondary sources. The predominant mechanism causing the accumulation of salt in the root zone of agricultural soils is loss of water through evapotranspiration (i.e., combined processes of evaporation from the soil surface and plant transpiration), which selectively removes water, leaving salts behind.

The accumulation of soil salinity can result in reduced plant growth, reduced yields, and, in severe cases, crop failure. Salinity limits plant water uptake by reducing the osmotic potential, making it more difficult for the plant to extract water. Salinity may also cause specific ion toxicity or upset the nutritional balance of plants. In addition, the salt composition of the soil solution influences the composition of cations on the exchange complex of soil particles, which influences soil permeability and tilth.

Irrigated agriculture, which accounts for 35–40% of the world's total food and fiber, is adversely affected by soil salinity on roughly half of all irrigated soils (totaling about 250 million ha), with over 20 million ha severely effected by salinity worldwide (1). As a result of the potential detrimental impacts of soil salinity accumulation, it is a crucial soil chemical property that is routinely measured and monitored.

LABORATORY METHODS OF SOIL SALINITY MEASUREMENT

Although a variety of instrumentation has been developed for the measurement of soil salinity in the field, the most common technique is still laboratory analysis of aqueous extracts of soil samples. In the laboratory, soil salinity is commonly determined from the measurement of the electrical conductivity (EC) of soil solution extracts. The current-carrying capacity of soil solution is proportional to the concentration of ions in the solution. Soil salinity is quantified in terms of the total concentration of the soluble salts as measured by the EC of the soil solution in dS m^{-1} (2).

To determine EC, measurements are made in a cell containing two electrodes of constant geometry and distance of separation (3). The soil solution is placed between the two electrodes. An electrical potential is imposed across the electrodes, and the resistance of the solution between the electrodes is measured. The measured conductance is a consequence of the solution's

salt concentration and the electrode geometry whose effects are embodied in a cell constant. At constant potential, the current is inversely proportional to the solution's resistance:

$$EC_T = k/R_T \quad (1)$$

where EC_T is the electrical conductivity of the solution in $dS m^{-1}$ at temperature T ($^{\circ}C$), k is the cell constant, and R_T is the measured resistance at temperature T . One $dS m^{-1}$ is equivalent to one $mS cm^{-1}$ and one $mmho cm^{-1}$, where $mmho cm^{-1}$ are the obsolete units of EC.

Partitioning of solutes over the three soil phases (i.e., gas, liquid, solid) is influenced by the soil/water ratio at which the extract is made, so the ratio must be standardized to obtain results that can be applied and interpreted universally. Customarily, soil salinity has been defined in terms of laboratory measurements of the EC of the saturation extract (EC_e) because it is impractical for routine purposes to extract soil water from samples at typical field water contents. One widely used technique is to obtain an extract by vacuum filtration of a saturated soil paste made with distilled water (4). Commonly used extract ratios other than a saturated soil paste are 1:1, 1:2, and 1:5 soil/water mixtures. However, extracts at these ratios only provide relative salinity because the soils are adjusted to unnaturally high water contents not found in the field. Soil salinity can also be determined from the measurement of the EC of the soil solution at some defined field water content (EC_w). An example would be the EC at field capacity, which represents the water content of soil 2–3 days after irrigation when free drainage is negligible. Theoretically, EC_w is the best index of soil salinity because it is the salinity actually experienced by the plant root. Nevertheless, EC_w has not been widely used for two reasons: (a) it varies over the irrigation cycle as the soil water content changes and (b) methods for obtaining soil solution samples are too labor and cost intensive to be practical for field-scale applications (5).

Soil solution extracts can be obtained by various means. For disturbed samples, soil solution can be obtained in the laboratory by displacement, compaction, centrifugation, molecular adsorption, and vacuum- or pressure-extraction methods. For undisturbed samples, EC_w can be determined either in the laboratory on a soil solution sample collected with a soil-solution extractor (see Fig. 1) or directly in the field using *in situ*, imbibing-type porous-matrix salinity sensors (see Fig. 2).

Electrolytic conductivity increases at a rate of approximately 1.9% per degree centigrade increase in temperature in the range of 15–35 $^{\circ}C$. Customarily, EC is expressed at a reference temperature of 25 $^{\circ}C$ for purposes of comparison. The EC (i.e., EC_e or EC_w) measured at a particular temperature T ($^{\circ}C$), EC_T ($dS m^{-1}$), can be adjusted to a reference EC at 25 $^{\circ}C$, EC_{25} (8):

$$EC_{25} = f_T \cdot EC_T \quad (2)$$

$$f_T = 0.4470 + 1.4034 \exp(-T/26.815) \quad (3)$$

where f_T is a temperature conversion factor.

Theoretical and empirical approaches are available to predict the EC of a solution from its solute composition (3).

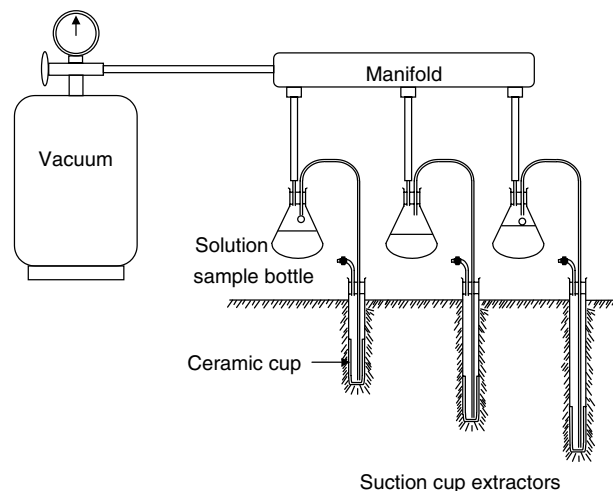


Figure 1. Diagram of a basic suction cup extractor setup for sampling the soil solution. Courtesy of Corwin (6).

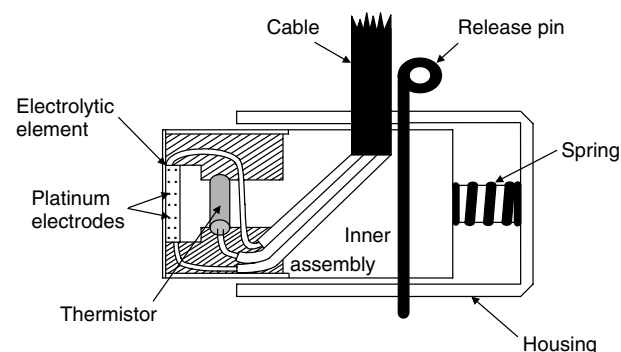


Figure 2. Schematic of a porous-matrix salinity sensor. Courtesy of Corwin (7).

Equation 4 is an example of a theoretical approach based on Kohlrausch's Law of independent migration of ions, where each ion contributes to the current-carrying ability of an electrolyte solution:

$$EC = \sum_i EC_i = \sum_i \frac{c_i(\lambda_0^i - \beta\sqrt{c_i})}{1000} \quad (4)$$

where EC is the specific conductance ($dS m^{-1}$), EC_i is the ionic specific conductance ($dS m^{-1}$), c_i is the concentration of the i th ion ($mmol_c L^{-1}$), λ_0^i is the ionic equivalent conductance at infinite dilution ($cm^2 S mol_c^{-1}$), and β is an empirical interactive parameter (9). Equation 5 shows an empirical equation developed by Marion and Babcock (10):

$$\log TSS = 0.990 + 1.055 \log EC \quad (r^2 = 0.993) \quad (5)$$

where TSS is the total soluble salt concentration ($mmol_c L^{-1}$).

FIELD METHODS OF SOIL SALINITY MEASUREMENT

Serious doubts exist about the ability of soil core samples, soil solution extractors, and porous-matrix salinity sensors (also known as soil salinity sensors) to provide representative soil water samples. Small

soil core samples, extractors, and salt sensors do not adequately integrate spatial variability because of their small sphere of measurement; consequently, Biggar and Nielsen (11) suggested that these are “point samples” that can provide qualitative measurement of soil solutions, but not quantitative measurements unless their field-scale variability is established.

As salinity is a highly spatially variable soil property, the use of EC_e to measure salinity at field scales is generally impractical because of the need for hundreds and even thousands of soil samples. The use of EC_e to measure salinity at field scales is only practical when soil sampling is directed using correlated spatial information. Two potential sources of correlated spatial information used to direct where soil samples should be taken to measure EC_e are: (a) visual crop observations and (b) geospatial measurements of apparent soil electrical conductivity (EC_a) using electrical resistivity (ER), electromagnetic induction (EMI), or time-domain reflectometry (TDR). These sources of spatial information are used to direct soil sampling to characterize field-scale soil salinity variation.

Visual Crop Observation

Visual crop observation is a quick and economical method, but salinity development is detected after crop damage has occurred. For this reason, visual observations are unrealistic because crop yields are reduced to obtain soil salinity information. However, remote imagery potentially represents a quantitative approach to visual observation. Remote imagery may offer a potential for early detection of the onset of salinity damage to plants.

Apparent Soil Electrical Conductivity

As a result of the time and cost of obtaining soil solution extracts, developments in the measurement of soil EC have shifted to the measurement of electrical conductivity of the bulk soil, referred to as the apparent soil electrical conductance (EC_a). Apparent soil electrical conductivity measures the conductance through not only the soil solution but through the solid soil particles and via exchangeable cations that exist at the solid-liquid interface of clay minerals. The techniques of ER, EMI, and TDR measure EC_a .

The interpretation of EC_a measurements is not trivial because of the complexity of current flow in the bulk soil. Three pathways of current flow contribute to the EC_a measurement: (a) a liquid phase pathway via salts contained in the soil water occupying the large pores, (b) a solid-liquid phase pathway primarily via exchangeable cations associated with clay minerals, and (c) a solid pathway via soil particles that are in direct and continuous contact with one another (5). As a result of the three pathways of conductance, the EC_a measurement is influenced by several soil physical and chemical properties: (a) soil salinity, (b) saturation percentage, (c) water content, and (d) bulk density. The saturation percentage and bulk density are both closely associated with the clay content. Measurements of EC_a as a measure of soil salinity must be interpreted with these influencing factors in mind. Traditionally, EC_e has been the standard measure of salinity used in all salt-tolerance

plant studies. As a result, a relation between EC_a and EC_e is needed to relate EC_a back to EC_e , which in turn is related to crop yield. In spite of these complicating factors, EC_a is the only viable means of spatially characterizing soil salinity at field scales.

As a result of the ease and quickness of measuring EC_a and the ability to mobilize EC_a -measurement equipment, geospatial measurements of EC_a are the most reliable means of characterizing the spatial variability of soil salinity at field scales. Geospatial measurements of EC_a are not used to directly measure soil salinity but rather to direct soil sampling from which EC_e measurements are subsequently made. From the geospatial EC_a measurements and a site-specific relation between EC_a and EC_e , a map of soil salinity distribution across the landscape and through the soil profile can be created. Detailed procedures and protocols for conducting an EC_a survey to characterize the spatial variability of soil salinity with EC_a -directed soil sampling are provided by Corwin and Lesch (12).

ER methods introduce an electrical current into the soil through current electrodes at the soil surface. The difference in current flow potential is measured at potential electrodes that are placed in the vicinity of the current flow (Fig. 3). The electrode configuration is referred to as a Wenner array when four electrodes are equidistantly spaced in a straight line at the soil surface. The two outer electrodes serve as the current or transmission electrodes, and the two inner electrodes serve as the potential or receiving electrodes. The depth of penetration of the electrical current and the volume of measurement increase as the interelectrode spacing, a , increases. For a homogeneous soil, the soil volume measured is roughly πa^3 , and the depth of penetration is roughly equal to a . Additional electrode configurations are discussed by Burger (14).

By mounting the electrodes to “fix” their spacing, considerable time for a measurement is saved. Veris Technologies¹ has developed a commercial mobile system for measuring and mapping EC_a using the principles of ER coupled to a global positioning system (GPS). The Veris Model 3100 consists of a pair of transmission coulter electrodes and a pair of potential coulter electrodes to measure the voltage drop. Two sets of arrays permit the measurement of EC_a to depths of 0–30 cm and 0–90 cm.

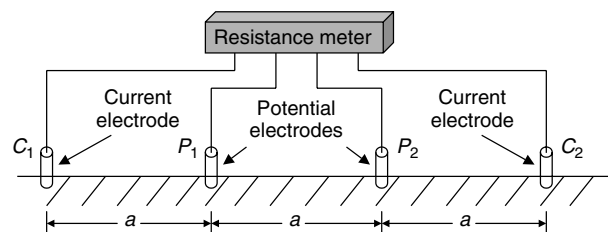


Figure 3. Schematic of electrical resistivity four electrodes, where C_1 and C_2 represent the current electrodes, P_1 and P_2 represent the potential electrodes, and a represents the interelectrode spacing. Courtesy of Corwin and Hendrickx (13).

¹Veris Technologies, Salina, Kansas, USA (www.veristech.com). Product identification is provided solely for the benefit of the reader and does not imply the endorsement of the USDA.

EMI instrumentation consists of a transmitter coil and a receiver coil (Fig. 4). The transmitter coil located at one end of the instrument induces circular eddy-current loops in the soil. The magnitude of these loops is directly proportional to the EC in the vicinity of that loop. Each current loop generates a secondary electromagnetic field that is proportional to the value of the current flowing within the loop. A fraction of the secondary induced electromagnetic field from each loop is intercepted by the receiver coil at the opposite end of the instrument, and the sum of these signals is amplified and formed into an output voltage, which is related to a depth-weighted EC_a . The amplitude and phase of the secondary field will differ from those of the primary field as a result of soil properties (e.g., clay content, water content, salinity), spacing and orientation of the coils, frequency, and distance from the soil surface (15).

The two most commonly used EMI conductivity meters in soil science are the Geonics² EM-31 and EM-38. The EM-38 has had considerably greater application for agricultural purposes because the depth of measurement corresponds roughly to the root zone (i.e., 1.5 m), when the instrument is placed in the vertical coil configuration. In the horizontal coil configuration, the depth of the measurement is 0.75–1.0 m. The operation of the EM-38 equipment is discussed in Hendrickx and Kachanoski (15).

Mobile EM equipment developed at the USDA-ARS Salinity Laboratory (16) is available for appraisal of soil salinity and other soil properties (e.g., water content and clay content) using an EM-38 or a dual-dipole EM-38 unit. The dual-dipole EM-38 conductivity meter simultaneously records data in both dipole orientations (horizontal and vertical). The mobile EM equipment is suited for the detailed mapping of EC_a and correlated soil properties at specified depth intervals through the root zone. The advantage of the mobile EM equipment is that it is noninvasive, so it can be used in dry, frozen, or stony soils that are not easily measured with the invasive “fixed-array” approach, which requires good electrode-soil contact. The disadvantage of the EM approach would be that the EC_a is a depth-weighted value, which is nonlinear with depth (17).

ER and EM are both well suited for field-scale applications because their volumes of measurement are large, which reduces the influence of local-scale variability.

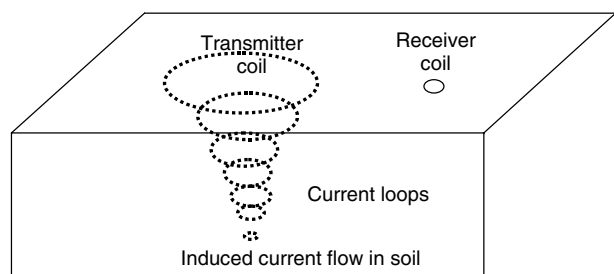


Figure 4. Transmitter coil, receiver coil, and induced current flow for EMI.

²Geonics Limited, Mississauga, Ontario, Canada. Product identification is provided solely for the benefit of the reader and does not imply the endorsement of the USDA.

However, ER is an invasive technique that requires good contact between the soil and four electrodes inserted into the soil; consequently, it produces less reliable measurements in dry, frozen, or stony soils than the noninvasive EM measurement. Nevertheless, ER has a flexibility that has proven advantageous for field application, i.e., the depth and volume of measurement can be easily changed by altering the spacing between the electrodes.

Time-domain reflectometry (TDR) was initially adapted for use in measuring water content. Later, Dalton et al. (18) demonstrated the utility of TDR to also measure EC_a , based on the attenuation of the applied signal voltage as it traverses the medium of interest. Although TDR has been demonstrated to compare closely with other accepted methods of EC_a measurement, it is still not sufficiently simple, robust, or fast enough for the general needs of field-scale soil salinity assessment (5). Only electrical resistivity and EM have been adapted for the geo-referenced measurement of EC_a at field scales and larger.

PLANT TOLERANCE TO SOIL SALINITY

The primary effect of soil salinity is on plant growth. Virtually all plant salt-tolerance studies conducted in the past have related plant yield response to EC_e , which is another reason why EC_e continues to be used as the primary measure of soil salinity. A salt-affected soil is one that contains enough soluble salts to hamper the growth of the crop. The proportions of dissolved ions (primarily Na^+ , Ca^{2+} , Mg^{2+} , Cl^- , SO_4^{2-} , and HCO_3^{3-}) can vary, and the effects on plants can vary. Generally, however, plants respond similarly to salinity over a fairly wide range of combinations of salt. To quantify yield response to soil salinity, Maas and Hoffman (19) proposed a two-piece linear response function with a tolerance plateau of slope zero and a concentration-dependent line whose slope indicates the yield reduction per unit increase in salinity:

$$Y_r = 100 - b (EC_e - a) \quad (6)$$

where a is the salinity threshold (dSm^{-1}), b is the slope expressed in percent per dSm^{-1} , and EC_e is the mean EC_e taken from the root zone. Values for a and b can be found in Maas (20).

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MAINTAINING SALT BALANCE ON IRRIGATED LAND

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DEFINITION

The salt balance of an area is described by an equation that compares the movement of salt into and out of an area. The salt fluxes are in balance when inflow and outflow of salt are equal. Salt moves in solution, and the water and salt balances are therefore linked as in Eq. 1, which describes the salt balance of an irrigated area:

$$IC_i + RC_r + GC_g = PC_p + \Delta S \quad (1)$$

where: I = irrigation water infiltrating the root zone

R = rainfall entering the root zone

G = capillary flow from the groundwater into the root zone

P = deep percolation from the root zone

C = salt concentration, and the subscripts referring to irrigation, rainfall, capillary flow, and percolation

ΔS = the change in salt content of the soil solution in the root zone

The water flows can be expressed in m³/ha/day or any other convenient set of units, such as mm/month, in which the unit of time indicates the period over which the salt balance is measured. The unit of salt concentration, C , is usually mg/L or part per million (ppm). Apart from areas close to the sea, it is reasonable to assume that C_r is zero. It is also often assumed that C_g equals C_p , which is reasonable for annual periods but not necessarily true for shorter periods.

For steady-state conditions, the salt concentration in the soil solution remains the same, and hence, ΔS equals zero. The ratio of the amounts of drainage water (percolating from the root zone) and the amount of water entering the root zone as irrigation and rainfall is called the leaching fraction, LF. Hence:

$$LF = P\Delta t / (I + R)\Delta t = P / (I + R) \quad (2)$$

where Δt equals the time period under consideration.

The ratio of the electrical conductivity (EC) of the drainage water to that of the applied water ($A = I + R$) can be written as EC_p/EC_a . For steady-state conditions, this ratio is proportional to the inverse of the leaching fraction ($1/LF$). For solutions consisting mainly of chloride salts, the proportionality constant is 1. For concentrated solutions, the proportionality constant is less than 1. Often irrigation water also has a proportionality constant of less than 1, and its value depends on the LF. For example, the proportionality constant of Colorado River water dropped to below 1 when LF became less than about 20% (1). The lower the value of LF, the lower the proportionality constant. Leaching fractions of less than 20% are common in irrigation practice. The relationship between EC_p/EC_a and $1/L$ is also influenced by the solubility of soil lime, which is a function of the partial pressure of CO₂.

PRESENCE OF SALTS

The salts carried in irrigation water come from salts that are continuously released from rocks and soils by mineral weathering and dissolution. Because of their low rainfall, most arid and semiarid lands are rich in these primary salts. The salts were deposited over time in sedimentary rock layers and are dissolved in the waters of aquifers. Unfavorable salt balances are common in arid lands: The global extent of salt-affected lands is considerable. It has been estimated that worldwide 100 million ha of irrigated land suffer from salinity, 20% of which is severely affected (2). Annually some 2 to 3 million ha of potentially productive agricultural land are taken out of production because of salinization. How much of this land is reclaimed (to various degrees) and then cultivated again is unknown. Wide divergence in the salinity figures is reported by different institutions, probably because of incomplete data and different definitions of salinity-affected lands; e.g., estimates for salt-affected land in India, according to different sources, range from 7 to 26 million ha, or between 17% and 60% of the irrigated land. For Pakistan, the most likely figure is some 40%, Israel 13%, Australia 20%, China 15%, Iraq 50%, and Egypt 35% of the irrigated land (3).

In many coastal regions, salt water intrusion has a strong negative effect on the regional salt balance. The problem is associated with groundwater extraction from deep fractured aquifers in the coastal region, often for drinking water in a coastal city. Sea water intrusion into the delta of rivers also upsets the regional salt balance. Here the cause is usually a much diminished outflow from the river because of upstream extractions for agricultural and industrial use. Examples include the Colorado River in the United States, the Murray in South Australia, and the Rhine in the Netherlands.

SALT BALANCE OF THE ROOT ZONE

Soluble salts added with the irrigation water must be removed by leaching to maintain the salt balance in the root zone of irrigated land. If not, crop yields will drop and—in case of predominantly sodium salts—soil crusting is likely to occur with an adverse effect on infiltration and hydraulic conductivity of the soil. The less soluble salts gradually precipitate when solutions become more concentrated and are again mobilized when the solutions become diluted by mixing with freshwater. Depending on whether precipitation or dissolution of salts (e.g., of lime or gypsum) occurs in the soil, the amount of salt leached from the root zone must be less than or more than that applied by irrigation. Generally, again depending on the occurrence of precipitation and dissolution reactions, when too much irrigation water is applied to the soil, more salts are added with the water, and in consequence, the drainage water also contains more salts than necessary. If the drainage water is returned to the river downstream, water quality becomes worse. Salt concentrations in the root zone must therefore be controlled to sustain irrigated agriculture over the long term.

When the quality of irrigation water deteriorates, for example, by extensive use of (slightly) saline groundwater when canal water became scarce, nonsteady conditions

occur and cation reactions will change the salt load of drainage water. Because of exchange of sodium and magnesium for calcium on the clay particles, mineral precipitation could double in a soil profile that was initially high in exchangeable calcium. These nonsteady conditions could last for many years before a new steady state evolves. Irrigation thus interferes with the geohydrological and geochemical regimes of the area. When irrigation was introduced in areas with low groundwater tables, excessive irrigation often occurred and deep percolation was the agent of interference. The main consequences of this increased groundwater recharge are a high watertable and the mobilization of natural salt residues of the semiarid lands. The latter is referred to as salt pickup.

The relationship among the salinity of the applied water, the leaching fraction, and the resulting soil salinity is an important one. Crop yields and the salinity of drainage water could be more accurately predicted if one could unambiguously calculate the soil salinity resulting from irrigation applications of known salinity and at a specified leaching fraction. Table 1 presents various relationships among leaching fraction (LF), the dimensionless ratio of the average weighted root-zone salinity (C_s), and the average salinity of applied water (C_a).

The values in the table are based on steady-state conditions. Because of differences between the underlying assumptions, considerable variation exists in the calculated ratios of root zone salinity to irrigation water salinity. For the four methods of Table 1, the salinity-tolerance data are from threshold salinity-response functions, and the leaching equations ignore the effect of sodium salts on the soil structure. The relation between soil and water salinity as governed by leaching is a dynamic one and hard to predict, subject as it is to site-specific feedback mechanisms among growth of the crop (hence, their water use through evapotranspiration), weather conditions (5), and the leaching of salts (6). A contributing factor is the variability in measured values of the EC of saturated-soil extracts. The coefficient of variation of the EC of soil moisture at saturation is about 50% (7).

Below a leaching fraction of 0.1, a small change in the leaching amount can make a large difference in root zone salinity. The ratio of root zone salinity to irrigation water salinity is less sensitive to changes in the leaching amount at LF values between 0.1 and 0.4, which are most common. In this range of LF values, root-zone salinity increases about linearly with the salinity of the applied water. With the introduction of precision irrigation to reduce water losses and reduce drainage flows, leaching fractions tend

Table 1. Relationships Between Leaching Fraction and Ratio of Soil Salinity Over Applied Water Salinity

LF	C_s/C_a^1	C_s/C_a^1	C_s/C_a^1	C_s/C_a^1
0.05	3.00	7.00	4.0	10.50
0.1	2.00	5.00	2.6	5.50
0.2	1.25	3.00	1.4	3.00
0.3	1.00	2.50	1.3	2.15
0.4	0.83	2.35	1.0	1.75

Source: Kijne (4).

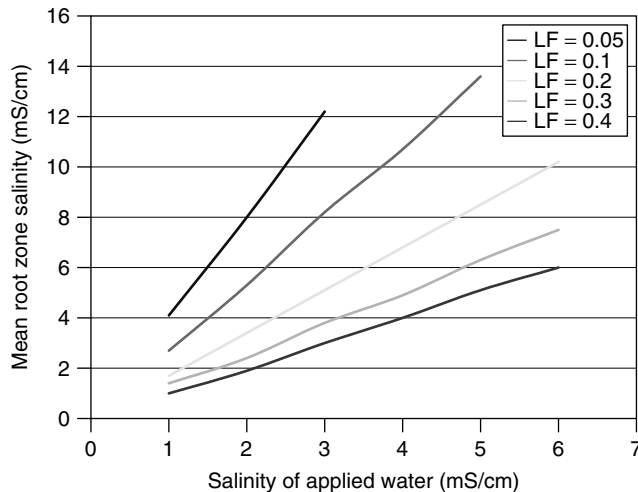


Figure 1. Root zone salinity as a function of the applied water and the leaching fraction. [Source: Hoffman and van Genuchten (8)].

to be lower with an enhanced risk of salt accumulation in the root zone.

The relationship of the fourth column in Table 1 uses a root water uptake function that is exponential with depth. This function is often used in modeling studies where a relationship between leaching and root-zone salinity is required. It is plotted in Fig. 1.

LEACHING OF SALT TO PREVENT YIELD REDUCTION

In irrigation practice, the leaching requirement is usually linked to the expected yield reduction. During leaching, soil salinity in the root zone increases with depth. In planning the desired leaching fraction for crop production purposes, it is commonly assumed that the EC at the lower root zone boundary corresponding to 25% to 50% yield reduction is still acceptable. The weighted average EC value for the entire root zone (weighted according to root distribution) would then be less than at the lower root zone boundary, and the resulting yield reduction is assumed to be less than 25% to 50%. This result gives some indication of the required leaching fraction based on the relation between yield and soil salinity.

The rate of downward flow is highest shortly after irrigation when the soil water content is still above or near field capacity. Thereafter leaching continues at a much-reduced rate. When the soil profile dries between irrigations, the soil solution becomes more concentrated. In many soils, the soil solution at field capacity is about twice as concentrated as when the soil is saturated. A further complication is that not all downward flow is equally effective in leaching salts from the root zone. Leaching is more effective when water moves through the soil mass, rather than through cracks between aggregates. Water moving through cracks and wormholes is called preferential flow. The efficiency of leaching depends on the fraction of irrigation water moving down the soil profile mixing with the soil solution. At the start of the leaching process as much as three-quarter of the applied water may flow through cracks in clay soils. Once the soil swells

up with moisture, cracks close and the leaching process becomes more efficient.

LEACHING CONTROL

Leaching practices influence the quality and quantity of drainage water. Their control is important when rising watertables are a problem (waterlogging) or where the salt load in the drainage water needs to be brought down. In such cases, restrictions are placed on the amount of salt farmers are allowed to discharge in the drainage water. Experience in Australia and the United States indicates that strict adherence to a set of site-specific regulations on the amount and quality of drainage flows and recharge to the groundwater is necessary to maintain the salt balance in irrigated agriculture. Because of the presence of monitoring systems, farmers in those countries can make well-informed choices about the crops to be grown and the cropping intensities they adopt. Thus, agricultural choices are made within a strict set of rules and the farmers are made responsible for the consequences of their decisions. The successful introduction of regulations with respect to permissible pumping from and recharge to the groundwater, including salt loading, in developing countries will be conditional on the development of adequate monitoring systems and management institutions.

RIVER SALINITY

All arid zone rivers have natural salt profiles resulting from mobilization of salts in the catchment area and saline seeps. Also, irrigation-induced transport contributes to river salinity as drain water with fossil salts from pumped groundwater discharges into a river. Increases in the salinity of rivers and streams in many dry parts of the world pose an ecological hazard that has largely been overlooked. This lack of attention for the ecological impact of increased salinity in inland waters is unfortunate considering the vulnerability of aquatic ecosystems to increased salt levels (9).

Salinity profiles for four rivers are shown in Fig. 2, which illustrates the various degrees to which salts are returned to the river or remain in the land and the groundwater (10). The data used in Fig. 2 are averages that ignore the seasonal changes in salt load as affected by the flow regime of the rivers. Most drainage water from agricultural land in Pakistan's Punjab is being reused, either from surface drains or pumped up from shallow groundwater. In fact, in some irrigation systems, one half to two thirds of the irrigation water is pumped from the groundwater. According to one estimate, about 116 million tons of salt are added annually to the irrigated land in Pakistan (11). Because of the absence of extensive drainage, the leached salts are returned to the land rather than disposed of in the Indus River system or in evaporation ponds. The average annual salt load of the Indus River water measured at rim stations in the catchment is estimated at 33 million tons, whereas the outflow to the sea contains only 16.4 million tons. Hence about half of the annual salt influx remains in the land

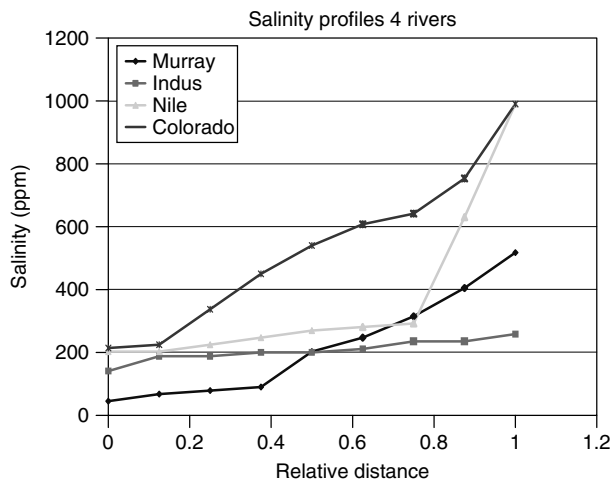


Figure 2. Salinity profiles of four rivers. [Source: Smedema (10)].

and in the groundwater (10). Most accumulation takes place in the Punjab, which is in sharp contrast with Egypt where a large portion of the irrigated land is underlain by subsurface drains that take the drainage water back to the Nile River. The salts do not stay in the Nile Basin but are discharged into the Mediterranean Sea. During part of the year, the salt content in the lower Indus is much lower than in the lower Nile and more salt disposal into the Indus could be accepted. However, during critically low flow periods, such disposals would not be possible as the salinity of the Indus water would exceed acceptable limits. The only option during those periods would be to store the drainage water temporarily for release during high flood periods. Extending the Left Bank Outfall Drain, now operating in Sindh, into Pakistan's Punjab may provide a more permanent, but also more expensive, solution than the present inadequate number of evaporation ponds.

The salt load of the Murray River shows considerable seasonal changes. At high flows when most of the river flow originates in the upper catchment, the salt influx is low, but at low flows of the river, when more of the river flow comes from groundwater base flow, the salt influx is higher. At low flows, salinity levels of the Murray are sometimes lowest downstream as the salinity of some tributaries is so much less than that of the main river that the water is diluted.

The salinity profile of the Colorado River is the result of large-scale irrigation development in the basin and concentration of the river water because of evaporative losses from storage reservoirs along its course. Under the salinity control projects of the Colorado River basin, several measures are undertaken to reduce further increases in river salinity. These measures include control of canal seepage by lining, restrictions on the quality of drainage flow back into the river, on-farm water savings by land leveling and changes in cropping patterns, retirement of irrigated land, and watershed management.

These examples of analyzing river salinity profiles clearly demonstrate their potential importance in understanding salt balances of irrigated lands at the scale of river basins.

MEASURING AND MODELING THE SALT BALANCE

Although the methodology for monitoring salinity at field level has been developed, using it in practice is problematic. The ability to diagnose and monitor field-scale salinity has improved considerably during the last 30 years or so. The apparent electrical conductivity of the soil can be measured by means of four-electrode sensors, either surface array or insertion probes, remote electromagnetic induction sensors (e.g., EM31 and EM38), and time-domain reflectometric sensors. Estimation of solute concentration from such measurements requires a calibration model relating soil solution EC to bulk EC. Many examples of the application and calibration of this technique illustrate the limited usefulness of manufacture-provided calibration curves and the different degrees of sophistication of the calibration models (12).

Various models have been developed to simulate transport through soils [e.g., LEACHM (13), SWAP (14)] that require soil solution EC data for their validation. These models are based on physical processes of water infiltration, redistribution, and evaporation. They enhance the understanding of water flow and chemical transport behavior and have been applied to simulate the terms of the water and salt balance (15). Widespread application of the models is restricted by the need for calibration with locally collected data and matching adjustment of some of the parameters. For example, in an Indian study with LEACHM-C, the root mean square errors (RSME) in the comparison of observed and predicted values of soil profile salinity ranged from 28% to 70%. After the water retention parameters were adjusted, the RMSE values ranged from 13% to 24% (16).

Similar problems of uncertainty in model application are associated with high watertables developing from excessive percolation in irrigated areas without adequate drainage. Plant roots take up water from shallow watertables as capillary rise meets part of the water demand of the crop. But if the shallow groundwater is saline, salt accumulation in the root zone affects crop production. Several models have been applied to this situation to ascertain the correct balance between the potentially positive and negative effects of shallow groundwater (17).

The spatial variability of soils and the temporal variability of the hydraulic parameters, especially in clay soils, as well as the largely unknown quantity of lateral groundwater flow constrain the applicability of models to larger areas. Yet, the focus of attention should be changed to the larger scale because of increased recycling of drainage water either through pumping from groundwater or by reusing water from surface drains. It is no longer sufficient to maintain a favorable salt balance at field scale; the salt balance should be maintained at system or river basin scale. As is apparent from the examples of river salinity mentioned earlier, the need for a holistic approach to the salt balance of irrigated areas is large and urgent.

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SALT TOLERANCE

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Salinity has adversely affected irrigated agriculture for thousands of years, probably since the first human civilization in Mesopotamia. The fall of this ancient civilization is attributed to both waterlogging and soil salinity (4). David Shedidan in his book “Desertification of the United States” calls soil salinization of irrigated lands one of the deadliest forms of desertification (5). No other region, where irrigated agriculture is practiced, can be an exception to this kind of desertification. In recent years, soil salinity has assumed a worldwide dimension and has become a serious threat to at least one-third of irrigated lands in the arid and semiarid regions where lots of irrigation water is used without providing adequate drainage. Although a long-term solution to the problem is to carry out improvements in land drainage, differences in salt tolerance of crops provide opportunities to continue growing increasingly salt-tolerant crops on salty lands to buy time before investments are arranged for reclamation.

SALT TOLERANCE: DEFINITION

The literal meaning of tolerance is to endure, sustain, or put up with. Thus, the salt tolerance of a plant could be defined as the plant's capacity to endure the effects of excess salt in the medium of root growth (1). That plants can withstand a fair amount of salts without adverse effects is implicit in this definition. In fact, a fair proportion of salts in the growing medium is needed for good growth. There are instances where plant growth is stimulated at low concentrations of salts, and only thereafter, the yield starts declining (2). Apparently, the problem of soil salinity develops only when the salts accumulate in the root zone to an extent that it becomes harmful to the plants. The salt tolerance of a plant is also expressed in terms of the yield decrease expected

for a given level of salinity compared to the yield under nonsaline conditions (3). Many times, salt tolerance and salt resistance have been used synonymously although, the two do not convey the same meaning. Salt resistance is a plant's ability to decrease or prevent the stress from penetrating into its tissues, but tolerance is the ability of the plant to eliminate, reduce, or repair the stress once it has already penetrated.

Salt or salinity in this article is used as a general term for salt stress, which could manifest itself from altogether different causes and compositions of salts under different situations. The stress could be due to bulk salinity of the soil water caused by excess soluble salts. Electrical conductivity (EC) of the saturation extract (EC_e) of a bulk soil solution has been widely used as a measure of this kind of stress (6). The adverse effects could also appear due to high exchangeable sodium, usually expressed in terms of exchangeable sodium percentage (ESP), due to excess ions such as chlorides and sodium expressed in $meq\ L^{-1}$, or due to excess toxic ions such as boron and fluorine, expressed in $mg\ L^{-1}$. A plant tolerant to one kind of stress need not necessarily be tolerant to other kinds of stresses. Therefore, a large number of relative tolerance tables are now available in the literature (1). These tables can provide a first approximation of the relative tolerance of plants to the anticipated kind and level of stress.

BIOLOGICAL RESPONSE TO SALINITY

The response of plants to salinity has been extensively studied and widely reported. For the purpose of discussions in this section, salinity effects are grouped under the heads (1) the effect of excess soluble salts, (2) the effect of high exchangeable sodium, and (3) a specific ion effect or toxicity of specific ions.

Effect of Excess Soluble Salts

Adverse effects appear to be related to the bulk soil salinity of the soil solution. As the salt concentration increases, the osmotic pressure of the soil water increases which affects the availability of water to the plants. When the osmotic pressure of the soil solution is high, the plants are unable to extract water as readily as they can from a relatively nonsaline soil. As such, even if water is present in the soil, it might not be available to the plants and results in physiological drought or osmotic desiccation. To survive, the plants must adjust osmotically and build up high internal solute concentrations, as is the case with salt accumulating halophytes. In salt sensitive plants, excess amounts of salt in the plant cells can be harmful because of damage to cell membrane, enzymes, and organelles.

The limited uptake of water hinders the uptake of ions by the plant root system. Thus, plants might encounter deficiencies of nutrients. Because the soil solution has high concentrations of certain ionic species, some of these ions, which are otherwise not harmful, might become toxic when absorbed in excess amounts. Salt-sensitive glycophytes have inadequate control over ion uptake, so such excess absorption of ions cannot be ruled out. Besides,

preferential absorption of one ion may also retard the absorption of other essential plant nutrients necessary for the normal growth of plants. Evidence of potassium deficiencies and inhibition of absorption of nitrate from saline soils can be cited.

Even glycophytes, which are salt tolerant and possess the ability to adjust osmotically, exhibit growth reduction largely on account of energy expenditure for uptake and synthesis of solutes.

Exchangeable Sodium

High ESP of a soil is associated with structural deterioration of soils. As a result, the edaphic environment is adversely affected and results in restricted air and water movement in the root zone. The factors that limit the uptake of water in alkali/sodic soils are a shallow wetting zone, an impermeable layer (crust) on the surface, low unsaturated hydraulic conductivity, and low available moisture content. Besides water availability, sodium toxicity, as a result of high ESP, can also affect the yield. High ESP can also disturb the mineral nutrition of the plant because it results in nutritionally inadequate amounts of Ca, or it might retard the absorption of some ions (7).

Specific Ion Effects

Some constituents of salts, for example, chlorides, which are harmless to most crops at moderate concentrations, may be toxic to some crops at higher concentrations, for example, citrus and grapes. In addition, some elements like lithium, and boron are toxic to plants even though they may be present in only very small quantities. Under such situations, plants would be subjected to toxic effects, nutritional imbalance, and deficiencies of certain essential elements.

In all the cases discussed in this section, it is believed that the adverse effects are usually due to the cumulative effects of several factors, although one may be dominating others in many cases.

DIFFERENCES IN SALT TOLERANCE OF PLANTS

Plants differ greatly in tolerance to salts. A wide range of salt-tolerant plants that evolved over millions of years exists in the plant kingdom. Plants with natural salt tolerance can survive in water of varying degrees of saltiness, from slightly brackish in the case of barley to ocean water in some hardy species of mangrove trees. Plant species called 'halophytes' are highly salt tolerant; some of them withstand salt concentrations in excess of that of seawater. At least 1500 of them are currently known (8), many of them have potential for commercial exploitation (9). These plants contain genes that protect them from salts in a variety of ways, such as growing salt-blocking membranes, excreting salt through their leaves, or sequestering salt in microscopic compartments called vacuoles. The natural vegetation of arid regions, xerophytes, also has high salt tolerance. On the other hand, almost all crop species are glycophytes, plants that do not grow well in the presence of salts. Most fruit

trees and vegetables are salt-sensitive plants. Generally, vegetable crops are more tolerant than fruit crops but are more sensitive to salt than field and forage crops. Within field and arable crops, beans, pulses, oil-seed crops, and cereal crops, in that order, show increasing sensitivity to salts. Many wild relatives of cultivated plants, however, are reportedly highly tolerant to soil salinity, alkalinity, and other associated conditions (10).

VISIBLE SYMPTOMS ON PLANTS GROWING UNDER HIGH SALINITY

Within a narrow range around a plant's salt tolerance level, no visible symptoms might appear on plants under field situations. Yet, the crop yield might still be affected. At high salinity, the plants show signs of wilting, a similar symptom that appears when a plant is water stressed. The leaves of salt stressed plants may turn yellow, a condition called chlorosis. It happens because the excess sodium blocks iron uptake and the lack of iron inhibits production of chlorophyll. Stunted growth and deep bluish green foliage of plants followed by low production are other common signs in plants grown under high salinity. The toxic effects that are characteristic of the specific ion causing toxicity usually show on plant parts.

FACTORS INFLUENCING SALT TOLERANCE

Absolute tolerance to salinity cannot be expressed explicitly. It varies considerably with edaphic and environmental factors, which influence salt tolerance in plants. The most important among the edaphic factors are the physical condition of the soil, soil fertility, salt species present in the soil solution, salt distribution in the profile, salinity level, and irrigation methods. The environmental factors include temperature, humidity, radiation, and pollutants in the environment. It has been found that poor aeration as a result of inadequate drainage may adversely affect plant response to soil salinity. For several crops, sensitivity to salts is known to differ markedly according to the type of salts present and as a result, isosmotic solutions of different composition might affect the plants differently. Sorghum plants exhibited greater growth reduction with sulfate type than chloride type salinity (11). Therefore, the type of salinity might alter the order of tolerance in crops. Since salinity stress is expressed as the total sum of the matric potential and the osmotic potential, the salt tolerance of crops would be significantly different in frequently irrigated than infrequently irrigated conditions. As such, shallow and frequent irrigations are recommended for crops growing under a saline/sodic environment. Because of the same reasoning, a plant response at the same salinity is better in drip irrigation than with surface irrigation. Gupta and Dubey (12) reported that irrigation water of much higher salinity could be used for growing vegetable crops with pitcher irrigation than with conventional surface irrigation techniques, as reported by Minhas and Gupta (13). Apparently, the salt tolerance of most crops under pressurized modern irrigation techniques needs to be worked out.

Plants respond better to salt stress in cool and humid than in hot and dry weather. Studies on several crops such as alfalfa, bean, beet, carrot, cotton, onion, squash, strawberry, clover, salt grass, and tomato have shown that salinity decreased yields more when these crops were grown at high temperatures (14–16). Hoffman et al. (17) observed that alfalfa yields were highest under ozone concentrations found in many agricultural areas in California at moderate salinity levels which were otherwise found to reduce growth in the absence of ozone.

Besides the environmental and edaphic factors discussed in the foregoing section, biological factors such as the varieties and rootstock of plants and the stage of growth also cause significant differences in the response of plants to salt stress. Evidence is abundant revealing inter- and intragenic differences in plants' response to salt stress. Green alga (*Dunaliella vindis*) survives in the highly saline Dead Sea at a salinity level exceeding 35 dS m^{-1} . A survey of reported data on the response of crop plants to increasing salinity levels reveals about eightfold differences among them in tolerance to salinity stress (18). An evaluation of more than 1000 wheat genotypes revealed a wide variation in the tolerance to soil sodicity (given by the pH of the soil solution). These wheat genotypes could be classified into tolerant, that grew well up to a pH of 9.6 (ESP and pH are well correlated for alluvial soils of the Indo-Gangetic plains) and sensitive, that did not grow well unless the pH was below 8.5. In between these two classes, varieties could be classified as medium tolerant and medium sensitive (19). Aswathappa et al. (20) recorded that some acacia species grew in NaCl concentrations of 1000 mol m^{-3} , whereas others stopped growing at 200 mol m^{-3} .

Some crops are known to be sensitive to salt stress at germination and others at emergence and early seedling growth. In cereal crops, the early productive stage has been found sensitive. Recent studies indicate that corn (21), sorghum (22), and cowpea (23) are most sensitive during the vegetative and early reproductive stages and least sensitive during the grain filling stage. It has been found that the tolerance of wheat increased with ontogeny. The soil saturation extract values at which yield would decline by 50% is 9.3 dS m^{-1} for seeding to crown root initiation stage and increases with age to 13.2 dS m^{-1} for dough stage to maturity (24). Though data sets are increasingly added to the literature to highlight such differences in response, some criticism is also appearing because of the limitations of the methods used in such evaluations. It is highlighted that presently used conventional methods of soil tests fail to reveal the true stress to which a plant is subjected at different growth stages. For example, it is difficult to say whether the young roots at the time of the emergence and initial establishment stage are really sensitive to salts or are exposed to a high salt stress particularly when the salt accumulate at the soil surface. Similarly, the salinity of the soil water that is approximately equal to that of the irrigation water at the top of the root zone is several times more than the salinity of irrigation water at the bottom. It is natural that at the later growth stages, plant roots moving deep into the profile might experience greater salt stress than that given

by the average soil salinity. Thus, one would tend to agree to a general statement that in many crops and plants, increased tolerance with age has been observed, yet the data are not sufficient to draw conclusions in absolute terms about the differences in salt tolerance at different growth stages.

CHARACTERIZATION AND ASSESSMENT OF SALT TOLERANCE

A number of models have been used to appraise the relative tolerance of crop plants. In some models, survival of the plants has been used as a criterion whereas in others, vegetative growth, absolute yield, and relative yield have been used to characterize relative salt tolerance. When comparisons are made among the models, some drawbacks are apparent in a few of them. For example, survival of the plants used in many ecological works is very often at the expense of production. Thus, it seems inappropriate for agricultural situations. Similarly, vegetative growth has a limitation. It has been seen that, where vegetative growth has been affected, many plants pick up later and yield almost at the optimum level. Absolute yield is useful agriculturally, as it tells the farmer what to expect, but has little use for quantifying differences in tolerance between crops with different levels of yield, for example, sugar cane and wheat. The use of relative yield compared to a control (say, non saline environment) allows comparisons between crops and cultivars, where the yield is expressed in different units or where the components of economic yield differ between them. More and more reports are now appearing where crop production functions are plotted in terms of relative yield as a continuous function of increasing salinity or for that matter ESP or the concentration of a particular ion. At this stage, it seems to be the most appropriate criterion for evaluating the relative salt tolerance of plants, although some limitations are known.

Maas and Hoffman (3), probably for the first time, proposed a linear response model, containing three independent parameters, the non saline control yield ($Y_{\max}$), a salinity threshold (EC_t), below which yield (Y) was unaffected, and a slope (S) representing the decrease in yield with increase in salinity above this level. EC_0 represents the salinity at which there is no yield. Mathematically, the response function is written as

$$Y = Y_{\max} \quad 0 \leq EC \leq EC_t \quad (1)$$

$$Y = Y_{\max} - S(EC - EC_t) \quad EC_t < EC \leq EC_0 \quad (2)$$

$$Y = 0 \quad EC > EC_0 \quad (3)$$

The relative yield of the crop as a ratio of the nonsaline control could also be used in this model. Then, Y would become RY , the relative yield and $Y_{\max}$ would be equal to 1 or 100 depending upon whether the relative yield is expressed in terms of a fraction or percentage.

The piecewise linear model has been used to produce tables of salinity response functions for a number of crops (1,3). Later studies have established that such a model can effectively describe the production function as

affected by high ESP (25). A model proposed independently by Oosterbaan is also based on the same premise and uses segmented regressions defining the threshold as a breakpoint. The model proposed by Oosterbaan (26) seems to be quite flexible and could be used for many more situations besides studying the salt tolerance of crops.

Based on this model, threshold salinity, the salinity at which a 50% (EC_{50}) reduction in yield is expected, and the salinity at which the yield is zero (EC_0) have been used to assess the relative salt tolerance of crops. Both the threshold salinity and EC_0 are not well defined (2), so EC_{50} seems to be the most appropriate parameter to assess the relative salt tolerance of crops (18,27), as it integrates both the slope and the threshold. The data derived from Maas (28) show that both EC_t and slope overlap between two classes (Table 1). Such overlap gets masked while calculating the EC_{50} . van Genuchten (2) proposed a sigmoid model, defined in terms of EC_{50} and p where again EC_{50} is the salinity where the yield is reduced by 50%, and p is an empirical constant defining the shape of the curve. This model also fits well with almost all the data sets of the Maas and Hoffman model, so it attests to the fact that EC_{50} could be a good parameter to define relative salt tolerance.

The use of this model in field experimentation, besides other limitations, may be difficult where differences between genotypes are small. A limitation of this model is explained well in Fig. 1. Using EC_{50} as the criterion, it would be concluded that the tolerance of four crops is in the order crop a > crop b = crop c > crop d. On the other hand, within crops b and c, one would tend to conclude that crop b is more tolerant to salt at low and medium salinity levels and crop c is more tolerant at high levels of salinity. Another problem with this model is that it can be used to identify tolerance as distinct from yield potential in many crop cultivars. In wheat, for instance, Kharchia 65, a wheat variety from Rajasthan (India) showed the highest degree of salt tolerance though it was not one of the top yielders amongst the varieties tested at Karnal (India). Notwithstanding such minor difficulties, this approach has the potential of application in categorizing the relative salt tolerance of crops.

RELATIVE SALT TOLERANCE OF CROPS

Extensive data on this issue are part of the literature. The most comprehensive information on this aspect is compiled and reported by Maas (1). This includes information on the relative salt tolerance of herbaceous crops (fiber, grains, grasses, forage, and special crops), vegetable and fruit crops, woody crops, ornamental shrubs, trees, and ground cover crops to the bulk soil salinity of the growing media. It

Table 1. Range of Model Parameters in Describing the Relative Salt Tolerance of Agricultural Crops

Relative Salt Tolerance of Crops	EC_t (dS m ⁻¹)	Slope (%)
Sensitive	1–2	15–35
Moderately sensitive	1–3	6–20
Moderately tolerant	3–7	4–20
Tolerant	4–8	4–7

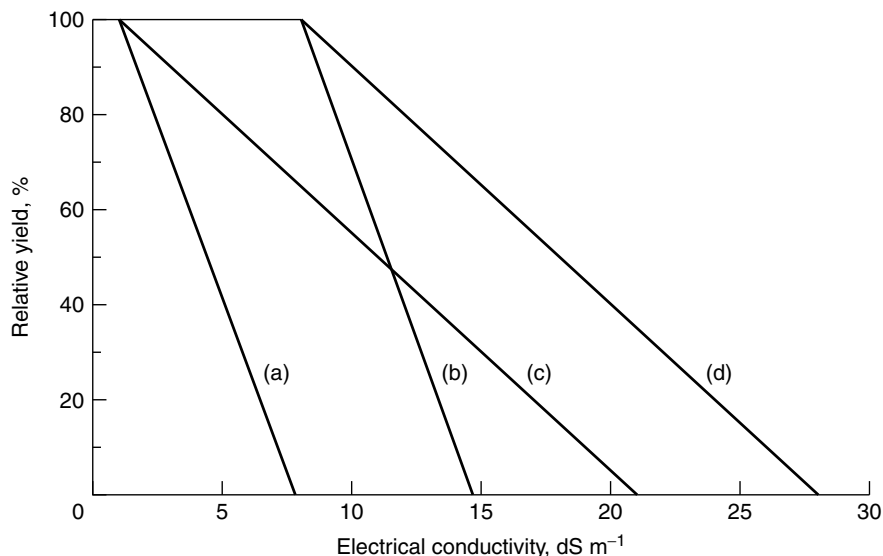


Figure 1. Crop salt tolerance as revealed by the piecewise linear model of Maas and Hoffman.

also lists the tolerance of agricultural crops and fruit-crop cultivars and rootstocks to chloride and the tolerance of agricultural crops and citrus and stone fruit rootstocks to boron. A number of tables listing the relative tolerance of crops have also appeared, but most are based on data reported by Maas and his co-workers. A table showing the relative tolerance of field, vegetable, forage, and fruit crops is given in Appendix 1. Gupta and Sharma (25) reported on the tolerance of a few agricultural crops and grasses to soil ESP. Porwal et al. (29) compiled the available data in India on tolerance of crops to saline water and presented the threshold and slope of the Maas and Hoffman function in terms of irrigation water quality. Per se, saline water might not directly affect crop yield, but once the leaching fractions under a given set of conditions are known, this data could be effectively used to assess crop tolerance to soil salinity. Later on, similar data were compiled and reported by Minhas and Gupta (13).

Although extensive literature is available on the relative tolerance to salts, considering the different agroclimatic conditions, different methods employed, varying farming conditions, and varieties used, the results obtained do not always agree. As an example, the alkali tolerance of rice in the literature has been rated as medium, but in the Indian context, it has been rated as highly tolerant. This difference arises because of the fact that the earlier rating was for direct seeded rice and the India rating is based upon transplanted rice where 35–40 day seedlings have been used for transplanting.

SALT TOLERANCE AND MANAGEMENT OF POOR QUALITY LAND AND WATER RESOURCES

Several treatments and management practices can reduce the salt level in soil. However, there are some situations where it is either not possible or not economically practical to attain desirably low soil salinity levels. In such cases, proper plant selection is one way to moderate yield reductions caused by excessive soil salinity. The ability of a

plant to grow on salty lands or to remove salt from such soil makes salt-tolerant plants potentially useful for restoring salt-damaged soils. Salt-tolerant plants can also provide farmers with greater flexibility in crop selection besides reducing the leaching requirement. The difference in the salt tolerance of plants is now being exploited both on experimental and field scales to use salty water resources for crop production and to dispose of them in an ecofriendly manner with reduced investments. It is achieved through sequential concentration of water. In such a system, following each cycle of use, the volume of water decreases, but its salinity increases. Thus, after each cycle, a more salt tolerant crop is chosen before finally disposing of the water in an evaporation tank or by some other means (Fig. 2). This kind of practice has also been referred as a multiple water use system in the literature. The quantity of water to be disposed of at this stage is much less, so the investment in an evaporation tank is significantly reduced.

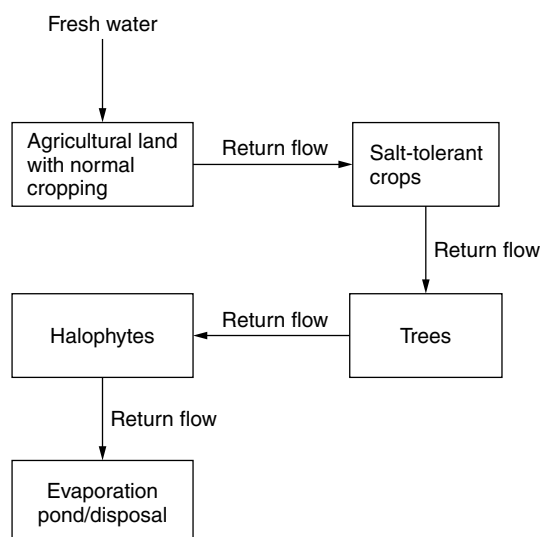


Figure 2. A schematic representation of sequential concentration for managing water resources.

RECENT DEVELOPMENTS IN SALT TOLERANT CROPS

Conventional breeding for developing salt-tolerant plants has been attempted for a long time, and the subject of breeding for salt tolerance has been reviewed by many workers (30). The lack of success in generating tolerant varieties [given the low number of varieties released and their limited salt tolerance (31)] would suggest that conventional breeding practices are not enough. Plant breeding from the existing gene pools could help to achieve small increases in salinity tolerance. Advances in molecular biology as well do not appear to have helped greatly in the development of salt-tolerant genotypes. There is good evidence that salt tolerance in truly tolerant species (halophytes) is a multifaceted trait. In halophytes, tolerance depends upon ion exclusion or compartmentation, synthesis of compatible solutes, regulation of transpiration, control of ion leakage through the apoplast, membrane characteristics, and the ability to tolerate low K:Na in the cytoplasm. It has been demonstrated that a range of traits is also important in crops. This suggests that the overall trait of salt tolerance is determined by a number of genes.

Twenty years ago, Epstein (32) argued for development of crops in which the consumable portion is botanically a fruit, such as grain, berries, or pomes, and that have a truly halophytic response to salinity. In these plants, Na⁺ ions would accumulate mainly in leaves, and because water transport to fruits and seeds is primarily through the phloem pathway (i.e., the intercellular connections), much of the salt from these organs would be screened. Initially, this hypothesis looked beyond realization. But in the last few years, scientists have reported increasing success in using genetic engineering to unravel the secrets of the genes and proteins that govern salt tolerance. A genetically engineered tomato that thrives in brackish water was produced in mid-2001 by a team of plant biologists at UC Davis (33). In late 2001, Purdue University scientists discovered a protein that guides salt into plants (34). Biotechnology specialist Alan McHughen at UC Riverside plans to begin research into producing salt-tolerant avocado trees. We also have other transgenic plants, for example, canola. Up to 10% of the dry weight of the leaves in this plant is sodium chloride. Schroeder and his colleagues have also discovered a gene that helps protect plants from salt damage by keeping salt in the roots. These results clearly support Epstein's argument.

With some success coming our way, it must not be forgotten that much more basic research is yet to be conducted. To develop genetically modified plants that have the triple characteristics of high yield, good quality, and tolerance to salts will not be an easy task in the foreseeable future. Besides, the genetically modified plants must be stable, so their descendants can be reliably grown with the added traits, and the plants must be tested for safety. This might take years of testing. Moreover, recent criticism and experience with genetically modified plants might hamper the progress of work. For example, some environmental groups say that large agricultural companies are rushing too quickly

to introduce genetically modified crops without regard for potential dangers. Genetically engineered corn to produce a natural insecticide has been opposed on the grounds that it could kill insects that are not pests, such as the monarch butterfly. Even more worrisome, early genetically modified plants weren't very useful to consumers. An earlier genetically engineered tomato called the Flav-Savr, also produced from research at UC Davis, flopped in the mid-1990s. To resolve such issues would require much more effort. Till then, an integrated approach based on conventional plant breeding, molecular biology, and genetic engineering should continue.

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APPENDIX A

Relative Tolerance of Crop/Plants to Soil Salinity (EC_e)^a

Tolerant	Moderately Tolerant	Sensitive
<i>Field Crops</i>		
Barley	Rye	Black gram
Sugar beet	Wheat	Bengal gram
Rape	Oat	Green gram
Cotton	Rice	Lentil
	Sorghum	
	Corn	
	Sunflower	
	Castor	
	Safflower	
	Soybean	
	Pearl millet	
	Linseed	
	Cluster bean	
	Pigeon pea	
	Cowpea	
	Sesame	
	Groundnut	
<i>Vegetable Crops</i>		
Garden beets	Bitter gourd	Radish
Asparagus	Bottle gourd	Carrot
Spinach	Brinjal	Celery
Amaranth	Tomato	Green beans
	Cabbage	Coriander
	Pea	Cumin
	Lady's finger	Mint
	Onion	
	Potato	
	Carrot	
	Turnip	
	Sweet potato	
	Dolichos	
	Sponge gourd	
	Watermelon	
	Muskmelon	
	Chillies	
	Fenugreek	
	Garlic	
<i>Forage Crops</i>		
Salt grass	Perennial rye grass	Meadow foxtail
Bermuda grass	Dallis grass	Alsike clover
Rhodes grass	Sudan grass	Red clover
Birdsfoot	Alfalfa	Burnet
Barley (hay)	Rye (hay)	
	Oats (hay)	
	Wheat (hay)	
	Orchard grass	
	Blue grass	
<i>Fruit Crops</i>		
Date palm	Pomegranate	Pear
	Guava	Apple
	Fig	Orange
	Olive	Plum
	Grape	Peach
	Ber	Mango
	Kagzi Lime	Avocado

^aThe values in the same column under a given category and for the type of crop/plant are in descending order of tolerance. The values are qualitative and are to be used as a first approximation.

GROUNDWATER ASSESSMENT USING SOIL SAMPLING TECHNIQUES

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Groundwater assessment for environmental projects involves significant soil sampling to evaluate hydrologic and lithologic conditions. Soil samples in environmental investigations are typically collected at a minimum of every one to two meters and at changes of lithology or obvious signs of contamination, based on soil staining or discoloration of the soil or in the case of volatile organic compounds, organic vapor readings. Continuous coring is recommended for a minimum of at least one soil boring per site to characterize the subsurface adequately.

VADOSE ZONE

Soil samples collected in the vadose zone above the water table are in the unsaturated zone. Groundwater is moving generally vertically, and some horizontal movement is caused by rootlets, faults, impermeable layers, or other conduits. Soil samples collected within the capillary fringe or below the groundwater table are in the saturated zone.

The capillary fringe moves up and down with seasonal variations of the groundwater table. Petroleum hydrocarbons move vertically from the source, such as an underground storage tank, through the vadose zone. As the hydrocarbons move vertically through the lithologic section, in general, the finer grained the soil, the greater the lateral spreading of the contaminant (Fig. 1).

$K_p = \text{Conc soil/conc gas}$ Soil zone	Unsaturated zone
$K_p = \text{Conc soil/conc gas}$ Intermediate zone	
$K_h = \text{Conc gas/conc water}$ Capillary fringe Water table	Saturated zone
$K_d = \text{Conc soil/conc water}$ Groundwater	

Figure 1. Equilibrium expressions for partitioning volatile organic compounds among aqueous, vapor, and sorbed phases.

CAPILLARY FRINGE ZONE

The capillary fringe, within the saturated zone, is the zone immediately above the water table, where water is drawn upward by capillary attraction. Due to irregularities in pore sizes among the grains of sediment, the capillary water does not rise to an even height above the water table but rather forms an irregular capillary fringe. The capillary fringe is higher in fine-grained soils such as a fine silt than in coarse grained soil such as a gravel due to greater tensions created by the smaller pore openings. The capillary fringe can be the zone of direct evaporation of groundwater if the water table is close enough to the surface (1).

The vertical movement during a complete hydrologic cycle can be a few centimeters, in other cases, it can be tens of meters, depending on the recharge of the water, the proximity of pumping wells, and other factors. This movement greatly affects the zone above the water table, which includes the capillary fringe zone, especially when it is impacted by light nonaqueous phase liquids (LNAPL), also called floating product or free-product.

When the free-product of a hydrocarbon fuel, such as gasoline, moves up and down through the hydrologic cycle, a smear zone develops in the zone above the water table. Some of this highly impacted soil zone (HISZ) is above the current capillary fringe zone and may exist below the top of the current groundwater table. Therefore, a HISZ equals the cumulative thicknesses of the migrating capillary fringe zones and frequently exists in areas where free-product or high concentrations of dissolved hydrocarbons have existed for a long period (Fig. 2).

For hydrocarbon fuels, which are lighter than water, the soil sample just above the water table is typically collected and analyzed. This sample may contain the highest concentration of contaminants within a soil boring. Sometimes, buried or submerged contaminants are trapped beneath the water table when water levels rise, leaving a former free-product zone below the water table (2). In these cases, soil samples collected within the groundwater zone would be appropriate (Fig. 3).

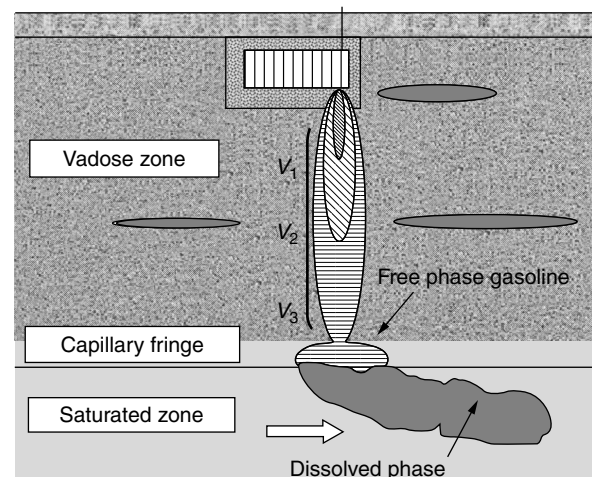


Figure 2. Gasoline movement from source (underground storage tank) to saturated zone (2).

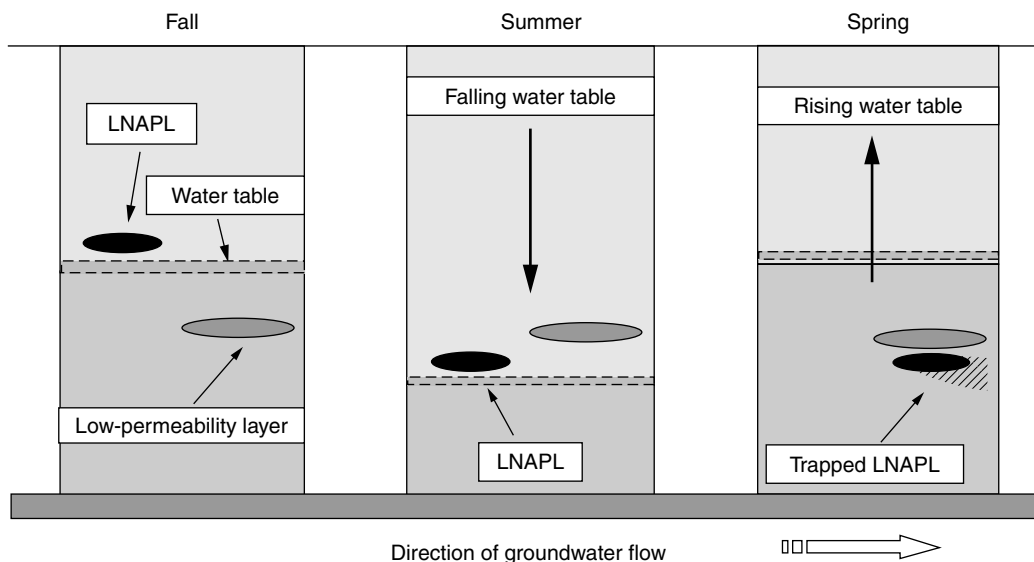


Figure 3. Trapping gasoline contaminants below the water table by fluctuating groundwater elevations (2).

For dense nonaqueous phase liquids (DNAPLs) such as chlorinated solvents, which have a specific gravity greater than water, soil samples are collected throughout the aquifer for hydrogeologic characterization. The highest concentrations of these dense liquids may be at the base of the aquifer, above the top of the next underlying aquitard.

SATURATED ZONE

Soil samples collected in the saturated or phreatic zone below the water table are used to evaluate the aquifer's characteristics. Soil samples collected below the groundwater table are generally not analyzed at a chemical laboratory, in favor of the more representative groundwater sample. Nonetheless, these saturated soil samples can provide a wealth of information about aquifer characteristics, total organic carbon, solid oxygen demand, porosity and permeability, as well as free-product distribution. Continuous soil coring of aquifer sediments allows complete lithologic analysis for potential groundwater remedial designs. These soil samples help scientists and engineers understand groundwater dynamics and the vertical and lateral movement of contaminants in the subsurface. Although groundwater sampling is imperative for groundwater assessments, data from soil samples used in conjunction with groundwater data provide a more complete understanding of subsurface conditions.

More detailed soil sampling information can be obtained from Testa (3), Jacobs (4), American Society for Testing Materials (5), California Regional Water Quality Control Board (6), and Morrison et al. (2). There are numerous minor differences in soil sampling and record keeping techniques and methods among field samplers, regional variations, and specific regulatory requirements that vary from state to state. A generalized summary of selected environmental soil sampling techniques follows.

SOIL SAMPLING METHODS

Undisturbed or minimally disturbed samples are obtained by driving a sampler into the soil, either manually, or using a drilling rig. The soil sampler is retrieved and opened, and the sample liner containing the soil sample is extracted. Samples are collected during subsurface investigations in vertical borings using rigs. Split-spoon samplers and direct push core samplers are used frequently for shallow soil sampling. The inner sample liners are typically composed of polyethylene, brass, or stainless steel. Samples may also be collected from tank pits and soil piles by driving a sampling tube into the soil by hand or using a drive-hammer.

Some analyses or guidelines allow collecting disturbed samples using a trowel, and then placing them in glass jars. Photoionization detectors (PID) are used in the field to measure organic vapors in soil samples. Soil samples should be collected in accordance with regulatory guidelines, which may vary among states and locales. The sample liners, containing soil, are typically sealed with Teflon tape and plastic caps at both ends. The samples are then labeled and stored in a refrigerated environment.

Small variations in soil lithology within several inches can lead to large differences in the results of chemical analysis. Duplicate soil samples, if taken, are collected as near as possible to the actual initial soil sample. Typically, these duplicate samples are collected in the same lithologic zone as the original sample. Duplicate samples should be a minimum of 5% of the samples. To maintain sample integrity, soil samples are delivered to certified laboratories using chain-of-custody procedures.

Lithologic Description

Soils and unconsolidated deposits for environmental projects are commonly described according to American Society for Testing Materials (ASTM) Method D 2488-84 (5) and the United Soil Classification System (USCS)

Major divisions	Group symbol	Description	Major divisions	Group symbol	Description
Coarse-grained soils More than half of materials is larger than no. 200 sieve size.	Gravels More than half of coarse fraction is larger than no. 4 sieve size.	GW	Fine-grained soils More than half of materials is smaller than no. 200 sieve size.	Silt and clays Liquid limit is less than 50	ML
		GP			CL
		GM			OL
		GC			MH
		SW			CH
	Sands more than half of coarse fraction is smaller than no. 4 sieve size.	SP			OH
		SM		Soils and clays Liquid limit is greater than 50	Pt
		SC			
			Highly organic soils		

Figure 4. Unified soil classification system (USCS).

for physical description and identification of soils. The USCS was first used in the geotechnical engineering field and is now commonly used in the environmental field. Unfortunately, the USCS is not very useful for describing lithologic characteristics, geologic properties, or the origin or depositional environment of the soil or sediments (Fig. 4).

Other soil identification systems include the Burmister Soil Identification System (BS) for unconsolidated deposits, which is commonly used with the USCS. The other soil description system is the Comprehensive Soil Classification System (CSCS) developed by the U.S. Department of Agriculture. The CSCS describes soils as to agricultural productivity potential and best agricultural land use.

The ASTM Soil Classification Flow Chart and the USCS are generally accepted soil description methods used in the engineering and environmental fields. Descriptions of moisture, density, strength, plasticity, etc. are made

using ASTM guidelines. Stratigraphic, genetic, and other data and interpretations are usually also recorded in the boring log. For consolidated deposits, including igneous, metamorphic, and sedimentary rocks, the USCS is combined with other geologic characteristics such as weathering, sorting, sphericity, and separation.

Color is commonly described by comparing the soil sample with a Munsell Rock Color Chart (7). Each layer of soil is identified by the geologist or engineer using the following items: color, soil type (USCS), classification symbol, color, consistency or relative density, moisture, structure (if any), and modifying information such as grain sizes, particle shape, cementation, plasticity, stratification, and other notes.

LITHOLOGIC INTERPRETATION

Once the lithology has been determined, geologists and engineers compare the data to develop a geologic model

for the depositional environment. Knowing the type and origin of the soil, sediments and rocks provides a better understanding of groundwater and contaminant movement and allows predictive modeling of an aquifer. Once the geochemical data have been evaluated, a site conceptual model can be developed, and a corrective action plan and remedial design can begin if the site requires environmental cleanup.

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SKIMMED GROUNDWATER

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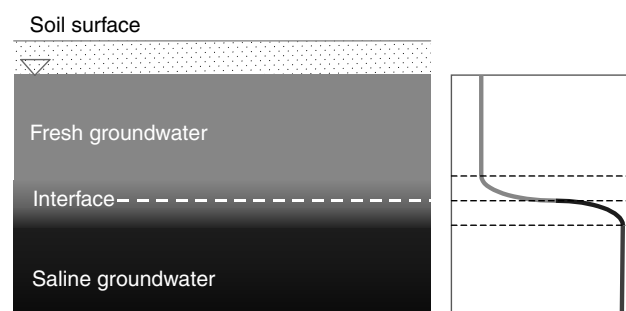
In several unconfined aquifers under irrigated agricultural areas, the groundwater system consists of a saturated porous medium containing miscible fluids (fresh groundwater and saline groundwater). The native groundwater is deep and saline because of the marine origin of the hydrogeologic formation. However, fresh groundwater lenses from deep percolation of the extensive water conveyance and distribution system, as well as from irrigation and rainfall, overlie this native saline groundwater.

According to Sufi et al. (1), over 20 BCM of fresh groundwater is being recharged annually in the Indus Basin of Pakistan.

The thickness of these fresh groundwater lenses varies from a few meters to more than 100 meters. For instance, in the Indus Basin of Pakistan, this is around 30 m thick in the lower or central parts of the *doabs* (area between two rivers), 60 m or more along the margins of the *doabs*, and approximately 150 m near the rivers and canals (2). When groundwater is extracted from these fresh groundwater lenses, it is called skimmed groundwater. Different types of

irrigation wells are available for extracting this skimmed groundwater (3).

Under natural conditions, the saline groundwater tends to remain separate from the overlying fresh groundwater. However, when waterbodies of variable solute concentrations are in contact, molecular diffusion causes mixing across the line of contact. Due to groundwater and saline water movement, a zone of dispersion, which is also known as the interface, forms between these two fluids. This interface, which varies in thickness, is not static but responsive to recharge and discharge mechanisms. Thus, the processes of diffusion and dispersion result in a transition zone where the salinity gradually changes from completely fresh to fully saline.



Therefore, the use of irrigation wells for skimming fresh groundwater may cause mixing of the 'interface' either within the well or in the aquifer. As a result, the irrigation well starts supplying skimmed groundwater of marginal quality. Therefore, the decision regarding the use of skimmed groundwater must consider the quality of groundwater with depth, the local hydrogeologic conditions of the aquifer, the socioeconomics of skimming well installation and operation, the discharge rate, the duration of pumping, and above all, the tolerable limits of skimmed groundwater quality for its intended uses (4).

Farmers are using this skimmed groundwater to fill the gap between the supply and demand of surface water in the conjunctive water use environment of the Indus Basin of Pakistan (5). This irrigation water resource has emerged as a formidable poverty reduction tool. Potentially, skimmed groundwater is a huge resource of irrigation water. In the Indus Basin of Pakistan alone, it has been estimated that nearly 200 billion cubic meter (BCM) of fresh groundwater is lying above salty groundwater (6), and covers about 30% of the irrigated area of Pakistan.

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SOIL MOISTURE MEASUREMENT—NEUTRON

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The water (liquid and/or vapor phase) contained in the soil above the water table is called soil moisture. Soil moisture is a function of precipitation, soil texture, porosity, the rate of soil moisture uptake by plants, and the rate of evapotranspiration. Measurements of soil moisture are needed for several purposes, including irrigation planning and scheduling, agronomic studies, flow and transport processes in the unsaturated zone, and geotechnical investigations.

Soil moisture can be measured by direct and indirect methods. Direct methods are those wherein water is removed from a soil sample by evaporation, leaching, or chemical reaction, and the amount removed is determined. Indirect methods involve measuring some property of the soil that is affected by soil moisture content or measuring a property of some object placed in the soil, which comes to water equilibrium with the soil. The indirect methods generally require calibration. They are suitable for quick and repetitive measurements and do not require destructive sampling as needed in direct gravimetric methods (1).

Although a variety of indirect techniques are available for measuring soil moisture content, they are neither equivalent nor applicable in all situations. Thus, choosing a suitable method and following a well-defined procedure are important for obtaining reproducible and reliable information. Various *in situ* recording sensors, water budget models, and experimental approaches are used for soil moisture measurement, including neutron probe, time domain reflectometry, electromagnetic induction, electron resistivity tomography, tensiometers, heat dissipation sensors, and thermocouple psychrometers. Modern methods of soil moisture measurement such as airborne radar and satellite imagery are still at the research and development stage.

The neutron probe also called the “neutron moisture meter” is a nondestructive, indirect method commonly used for repetitive field measurement of soil water content volumetrically.

PRINCIPLE

The neutron probe is based on the principle of neutron thermalization. Hydrogen nuclei are nearly the same size and mass as neutrons and therefore, can scatter and slow neutrons. High-energy (0.1–10 Mev.), fast (1600 km/s) neutrons emitted from a radioactive source, such as americium–beryllium or radium–beryllium are slowed and changed in direction by elastic collisions with atomic nuclei of hydrogen atoms in soil. This process is called “thermalization,” and the low-energy (about 0.03 eV) neutrons, after collision, are called “thermal/thermalized neutrons.” The density of the thermal neutrons (thermal cloud of neutrons) depends on the soil–water content; the vast majority of hydrogen in soil is associated with water. Higher water content in soil leads to increased thermalization and, thus, a denser thermal cloud. A “slow neutron” detector is installed adjacent to the source (emitter) to measure the cloud density. The measurement is usually in the form of a “count ratio”; a higher count ratio denotes higher water content and vice versa. Thus, the density of thermal neutrons can be calibrated against the water content volumetric. (2).

THE EQUIPMENT

Figure 1 shows a typical neutron moisture meter. The equipment consists of

1. a neutron moisture depth probe containing a radioactive source that emits high-energy neutrons, and a detector of thermalized neutrons.
2. protective shielding (for gamma rays) of lead and polyethylene or paraffin for neutron absorption. The depth probe is stored within the shield when not in use.
3. a meter assembly that includes a pulse counter to register the counts generated by the thermal neutron detector, a built-in computer, a display screen, and a keyboard.
4. a thin-walled aluminum or plastic access tube. The tube diameter is recommended by manufacturers for optimized equipment performance.
5. a radioisotope license, radioisotope leak test kit, and radiation monitoring equipment, as required by the license.
6. calibration curves and parametric data for calibrating the computer of the equipment, if needed.

CALIBRATION OF EQUIPMENT

The sources of hydrogen other than water (the primary source of hydrogen) in a given soil are assumed constant and are accounted for during calibration. Similarly,

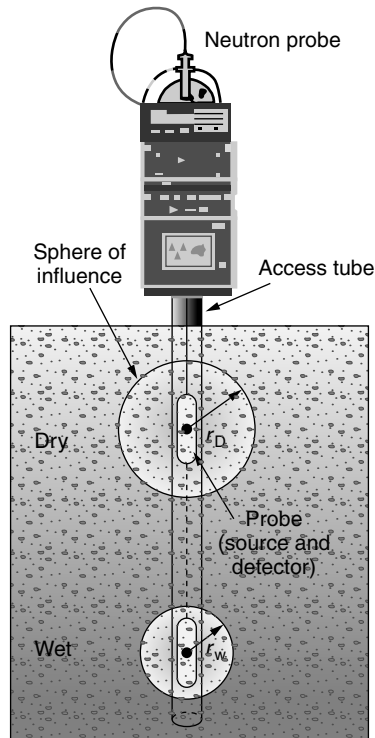
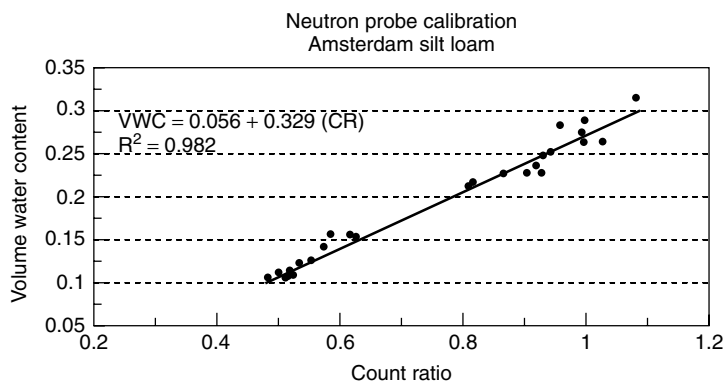


Figure 1. Neutron probe device. r_D and r_W are the radii of measurements in dry and wet soils (Source: Reference 3).

several nonhydrogen substances including C, Cd, B, Cl, and Li may be present in some soils and may also thermalize fast neutrons. However, their effect is compensated for through soil specific calibration. The neutron probe is calibrated by paired measurements of soil moisture content and neutron probe counts. Usually, the volumetric moisture content, θ_v is linearly related to the slow neutron count ratio (CR) as follows:

$$\theta_v = a + b(\text{CR})$$

where CR is the ratio of slow neutron counts at a specific location in the soil and a standard count obtained with the probe in its shield. A calibration curve can be obtained with a sufficient number of measurements. This calibration curve is approximately the same for many soils. Figure 2 depicts a typical calibration curve.



The sphere of influence of neutrons around the radiation source varies with water content from about 0.15 m for wet soil (almost saturated) to more than 0.60 m for very dry soil, depending on how far fast neutrons must travel to collide with a requisite number of hydrogen nuclei. An approximate relationship for the radius of influence, r (in cm), as a function of soil water content is given by

$$r = 15(\theta_v)^{-1/3}$$

PROCEDURE

1. Use the soil auger to form a hole for installing the access tube. The access tube is left so as to protrude about 10 cm above the soil surface and is covered with an empty can or a stopper to keep water and debris from entering.
2. Place the probe unit over the access tube to make a measurement. Select an appropriate counting time, and make several standard counts while the probe is in the shield. This will correct for any electronic changes in the counting circuits.
3. Make one or more counts at each selected measurement depth.
4. Use the calibration curve to convert the count ratios to volumetric moisture content, or read the moisture content directly if the equipment has a built-in computer.

COMMENTS AND PRECAUTIONS

1. In regard to the radiation hazard of devices with radioactive sources, it is strongly recommended that the equipment be used according to the manufacturer's instructions, which must comply with safe procedures dictated by the radioactive source license.
2. Site-specific calibration may be required for a soil of unusual neutron-absorption properties or if the diameter or material of the access tube is different from the manufacturer's recommendations.
3. As the zone of measurement is spherical and at least greater than 0.15 m in radius, the probe depth increment in the access tube should not be less than

Figure 2. A neutron probe calibration relationship (Source: Reference 3).

0.15 m. The approach to the soil surface should be no closer than 0.15 m because a portion of the neutrons may escape from the soil.

ADVANTAGES AND DISADVANTAGES

The neutron probe moisture meter has several advantages:

1. It is a widely accepted method of soil moisture measurement with highly reproducible results.
2. After initial tube installation, no further monitoring is required.
3. Both horizontal and vertical installation are possible.
4. It can measure soil water content at multiple depths and locations.
5. It is possible to couple the equipment with other devices.
6. The equipment produces reliable results for a long period.

Following are the disadvantages of the equipment:

1. It contains a source of radioactivity and, therefore, cannot be automated. It may also pose a health risk if adequate precautions are not taken.
2. The equipment is expensive and requires calibration.
3. It has relatively limited spatial resolution of measurement.

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SOIL N MANAGEMENT IMPACT ON THE QUALITY OF SURFACE AND SUBSURFACE WATER

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Nitrogen (N) in the forms of nitrate or ammonium is a nutrient needed for plant growth. When the amount of soil-supplied N is considered deficient for satisfactory crop yields, N fertilizer is added to supplement the soil-supplied N. If N is supplied in excess of crop needs, a surplus is created that can cause significant water

quality problems. Until recently, the main reason for investigations into the forms of N in soil systems below the rooting depth of plants was to evaluate the loss of nitrate to crops and to estimate the loss of production. In recent years, much attention has been focused on forms of N as potential pollutant sources of both groundwater and surface water. Nitrate concentrations above a certain level in drinking water may have critical health effects on animals and human infants. Alternately, N can also contribute to surface water contamination via surface water runoff and erosion. Excessive nitrate concentrations in surface water bodies, such as streams or lakes, can accelerate algae and plant growth, resulting in oxygen depletion. Nitrogen must be managed properly to protect water quality. Best management practices (BMPs) are methods that are the most effective, practical means of preventing or reducing nutrient pollution from nonpoint sources. Nitrogen BMPs primarily involve a change in fertilizer management practices, such as changing the timing, placement method, and/or the amount of fertilizer applied to reduce the possibilities of contaminating water resources. In addition, conservation tillage practices, cover crops, and crop rotation all help reduce surface runoff, erosion, and leaching of soil N. It is imperative to continue the development and implementation of new and effective management practices that increase N recovery and reduce potential N losses to the environment.

INTRODUCTION

Nitrogen is one of the most abundant elements on earth. It exhibits a number of oxidation states and combines with a number of other elements to form organic and inorganic compounds (1). Nitrogen accounts for 78% of the earth's atmosphere as elemental nitrogen (N₂) gas (2). However, the majority of the earth's N (98%) is in rock, sediment, and soils. The amount of N in rocks is about fifty times more than that in the atmosphere (3). For mineral soils in the United States, the approximate range of total N content in the surface six inches is 0.02% to 0.50% (4). Growing plants, animals, and microbial populations need a continual source of N. It is an essential component of the proteins that build cell material and plant tissue. In addition, it is necessary for the function of other biochemical agents, including chlorophyll (which makes photosynthesis possible), enzymes (which help organisms carry out biochemical processes and assimilate nutrients, and nucleic acids such as DNA and RNA which are involved in reproduction) (5). Of all the major plant nutrients, N is often the most important determinant of plant growth and crop yield. Without adequate N, plant growth is restricted. Characteristic symptoms of N deficiency in plants include yellowing and death of leaves and stunted growth. Just as too little N can cause problems, too much N can also cause problems. These problems can extend to plants, humans, animals, and the environment. The problems posed to the environment occur when excess N in soils is carried away with surface runoff to lakes, rivers, groundwater, and other aquatic ecosystems. Excess N in rivers, lakes, and groundwater can be toxic to humans and causes

water quality problems in natural water systems (6). Nitrogen, particularly as nitrate, is the most common contaminant in aquifer systems (7). Hallberg (8) points to agriculture as the most substantial anthropogenic source of nitrate, and Keeney (9) suggests that this is caused by intensive and extensive land use for crop and animal production. The United States Environmental Protection Agency (USEPA) has identified eutrophication as the main problem in U.S. surface waters that have impaired water quality (10). Excessive amounts of nutrients, especially N and phosphorus (P), speed up eutrophication (11). Excess N in the estuaries of the oceans enhances the growth of aquatic organisms to the point that they affect water quality and lower dissolved oxygen levels (12–14). This affects the metabolism and growth of oxygen requiring species, causing hypoxia (12,13). Implementation of N-BMPs (best management practices) allows efficient, profitable production and minimizes N movement to surface water and groundwater. This article addresses the concept of N movement in soils and the impacts of N management on surface and subsurface water quality.

SOURCES OF NITROGEN IN THE ENVIRONMENT

Atmosphere

The earth's atmosphere is made up of 78% N as a colorless, odorless, nontoxic gas. The same N_2 gas found in the atmosphere can be found in spaces between soil particles. However, plants cannot use this form of N (15). A group of microbes (*Rhizobium* bacteria) that naturally exists in the soil converts N to ammonium. In *biological fixation*, nodule-forming *Rhizobium* bacteria inhabit the roots of leguminous plants and through a symbiotic relationship, convert atmospheric N_2 to a form the plant can use (16).

Atmospheric fixation occurs during lightning storms. The enormous energy of lightning breaks down N_2 molecules and enables their atoms to combine with oxygen in the air to form nitrogen oxides. These dissolve in rain forming nitrates that are carried to the earth. Atmospheric N fixation probably contributes some 5–8% of the total N fixed (5).

The large amount of fossil fuels used to produce electricity and in manufacturing industries oxidizes N because the intense heat given off by their combustion causes the molecules to react with each other, thereby adding N to the air as nitrates. These nitrates fall to the earth during precipitation, adding more N to the soil (17).

Animal Manure

Animal manures are important sources of N in the terrestrial environment. Manure usually contains both organic and inorganic N forms. The amount of N in manure depends on the animal's species, age, diet, and bedding materials. Ammonium N is the primary inorganic form in manure and is readily available to crops. Organic N becomes available to plants as manure decomposes; 20 to 50% of organic N is available to the first crop after application. Much of the remaining organic N becomes available in subsequent years as mineralization continues (18).

Legumes

Leguminous crops, such as alfalfa, beans, and peas, generally do not require N fertilization because the *Rhizobium* bacteria living on their roots take N from the air and convert it to forms that plants can use. Legumes provide some of the N for their own growth and often build up a supply that can be used by crops that follow (19).

Commercial Fertilizers

Commercial fertilizers are derived from the atmospheric N pool. Industrial processes used to manufacture commercial fertilizers usually combine atmospheric N with hydrogen and/or oxygen and/or carbon and then convert the products to various N fertilizers such as ammonium nitrate, anhydrous ammonia, and urea (4). These fertilizers are generally applied directly to soils.

Crop Residues

Crop residues returned to the soil are important sources of N for succeeding crops if they are properly managed (20). Nitrogen exists in crop residues in complex organic forms; the residues are decayed by soil microbes, thereby converting N and other nutrients to inorganic forms, which then become available for plant uptake (16).

Organic Soil Matter

Organic soil matter is a major source of N in terrestrial ecosystems. Organic matter is composed primarily of a relatively stable material called humus. Even though the soil has a large pool of N stored in soil humus, this N is mostly insoluble and is not available for plant uptake. Organic N must be mineralized to inorganic forms by soil microbes before plant uptake can occur.

FORMS OF NITROGEN IN SOILS

Nitrogen in soils is found in organic and inorganic forms, both of which are important sources of N for plant and microbial uptake. Organic N occurs in crop residues and organic matter as protein compounds, amino acids, amino sugars, and complex, largely unidentified substances (4). More than 90% of all soil N exists as organic N, and for the most part, is not available for plant uptake. Natural biological processes carried out by microorganisms in soil convert organic N to inorganic forms, which plants can use (15). Inorganic N exists in soil primarily in three forms: elemental N, ammonium N (NH_4^+) and nitrate N (NO_3^-). Ammonium N and nitrate N are two forms of soil N that can be readily used by plants, whereas elemental N cannot be used directly by plants. However, some N-fixing bacteria, particularly *Rhizobium* that coexists with leguminous plants, use N in the air (elemental N) and converts it in to a form of N that can be used by plants.

NITROGEN LOSSES FROM SOILS

Nitrogen is lost from soil in several ways: leaching, denitrification, volatilization, soil erosion and runoff, and crop removal.

Leaching

Leaching is the loss of soluble N. The NO_3^- ion is negatively charged and is not attracted to soil's clay and humus, which are also negatively charged. So it stays in the water between soil particles. As water moves through soil, the nitrate in the soil solution moves along with the water (21). This loss most frequently occurs in areas of heavy rainfall, under excessive irrigation, and in coarsely textured soils.

Denitrification

Denitrification is the conversion of NO_3^- molecules to gaseous forms that are lost to the environment. When there is a deficit of oxygen in the soil (i.e., under anaerobic conditions), certain bacteria meet their energy needs by reducing NO_3^- to dinitrogen gas or to nitrogen oxide (N_2O). These gases are released to the atmosphere, resulting in a loss of N from soil (22). Denitrification can be significant when soils are saturated with water.

Volatilization

Volatilization is the loss of N from the soil as ammonia (NH_3) gas. NH_3 volatilization commonly takes place when N is in an organic form called urea. Urea may originate from animal manure, fertilizers and, to a lesser degree, from the decay of plant materials. Urea is converted to ammonium carbonate by urease, an enzyme found in soils. Ammonium carbonate easily dissociates to ammonium and carbonate ions. The carbonate ions result in a relatively short-lived increase in soil pH. Ammonium ions convert at high pH to NH_3 gas, which may escape to the air (23). This results in a net loss of N from the soil system.

Surface Runoff and Soil Erosion:

Surface runoff and soil erosion can also cause significant losses of soil N. Runoff and erosion losses include both inorganic and organic N. Erosion leads to the removal of organic N in the upper soil profile (24) and NH_4^+ N, which is adsorbed to soil particles (25). Nitrate N is water soluble and is generally not adsorbed to soil particles; hence, it quickly moves off the soil surface as runoff with increased rainfall.

Crop Removal/Harvest

Substantial amounts of N are lost from soil through crop removal (16). When crops are harvested, the N source is removed. Crop removal accounts for a major portion of the N that leaves the soil system.

HUMAN HEALTH AND WATER QUALITY CONCERNS RELATED TO LOADINGS OF N INTO SURFACE AND GROUNDWATER

Nitrogen is the one element of most concern as it relates to water quality. Although N can be found in many forms in the soil, the nitrate form primarily affects water quality (15). The contamination of surface water or groundwater by NO_3^- N presents a potential health

threat to human population. High NO_3^- N levels in water are a concern because the ingestion of such water can reduce the oxygen carrying capacity of the blood. Nitrate is reduced to NO_2^- , which oxidizes the Fe of the hemoglobin in the blood, and methemoglobin, which cannot carry oxygen, is formed (9). Infants have greater potential than adults to suffer these ill effects. In the United States, the maximum NO_3^- N level for potable water has been set at 10 mg/L (26). Secondary and related deleterious effects of exposure to NO_3^- include increased respiratory infections, inhibition of iodine uptake by the thyroid, and possible reproductive problems (27).

Another area of concern is the toxic effect of ammonia (NH_3) on freshwater aquatic life (28). It has been known since the early 1900s that NH_3 is toxic to fish and that this effect varies with water pH and temperature. A concentration of 0.02 mg/L as unionized NH_3 is the current limit in freshwater in the United States (29).

The most significant water quality concern with respect to N is the overenrichment or eutrophication of surface waters. Nitrates stimulate the growth of algae and other plankton, which provide food for higher organisms; however, excess NO_3^- N can cause over production of plankton. As these planktons die and decompose, they use up the oxygen dissolved in water, which causes an oxygen shortage for other aquatic organisms. Elevated N concentrations have altered natural aquatic floral and faunal population dynamics, exacerbated occurrences of hypoxia and anoxia, and sped eutrophication in the Gulf of Mexico (12,30).

NITROGEN BEST MANAGEMENT PRACTICES: IMPACT ON SURFACE AND GROUNDWATER QUALITY

Nitrogen from fertilizers as well as from other sources (e.g., biosolids, animal manures) has a direct impact on water quality. The primary goal of N best management practices (N-BMPs) is to manage N inputs to crop production to prevent degradation of water resources while maintaining profitability. Typically, BMPs are farming methods that assure optimum plant growth and minimize adverse environmental effects (31). By putting N-BMPs into practice, N losses from agricultural soils can be controlled. There are many BMPs currently in use; some of the more popular soil N management practices and their potential effects on surface and subsurface water quality have been categorized by Owens (32) as follows.

Crop Management

Cover Crops. The use of cover crops in cropping systems provides several benefits. However, only in recent years has much of the attention been focused on their potential to reduce NO_3^- leaching losses (32). The key to reducing NO_3^- N losses is to keep soil nitrate concentrations low during periods of high leaching potential. Nitrate can leach to groundwater when the soil contains significant NO_3^- N and when water percolates below the root zone. These conditions often exist during the winter months, when percolation of winter rainfall can be high and uptake of water and nitrate by plants is minimal (33).

Cover crops influence nitrate leaching in two ways: water budget effects and N uptake effects. Cover crops can influence the water budget by reducing the water content of the soil during their active growth stage (33,34). In humid climates, this water usage can reduce nitrate leaching significantly. Cover crops function by accumulating the inorganic soil N between main crop seasons and by holding it in an organic form, thus preventing it from leaching. Nonleguminous cover crops can immobilize as much as 70% of the available NO_3^- N in the upper soil profile (35). Generally, nonlegumes are about three times more efficient than legumes in reducing nitrate leaching (36). Meisinger et al. (33) reported that cover crops reduced both the mass of N leached and the NO_3^- -N concentration of leachate by 20–80% compared to no cover crop control. They also determined that grasses and *Brassicas* were two to three times more effective than legumes in reducing NO_3^- -N leaching. Because, grasses and grains become quickly established in the fall and develop an extensive root system, they are more efficient than legumes in capturing soil nitrate and preventing late fall and winter leaching to groundwater (35,37). In a western Oregon study of nitrate movement in the soil profile following harvest of a broccoli crop fertilized with 250 lb N/acre, Hemphill and Hart (38) demonstrated the effectiveness of a fall-planted rye cover crop in reducing the movement of soil nitrate to the soil profile. Areas with cover crop contained 0.18 mg/kg nitrate at 30–40 inches of soil depth compared to 5.38 mg/kg nitrate at the same depth where no cover crop was grown.

Controlling erosion is an important component of surface water quality protection. Cover crops protect against soil erosion (39,40). Cover crops can be effective in reducing runoff and erosion, especially during the winter when there is little crop residue (32). Farmland is most susceptible to erosion when there is no vegetative ground cover or plant residue on the soil surface. Cover crops provide vegetative cover when a crop is not present to cushion the force of falling raindrops, which otherwise would detach soil particles and make them more prone to erosion. The rate of runoff is also slowed, thus improving moisture infiltration into the soil (41). When legumes are used as the winter cover crop, fixed N in the legumes can be supplied to the subsequent crop so that less fertilizer N has to be applied. This reduces the amount of inorganic N, which is readily available for removal in surface runoff (32).

Crop Rotations. Understanding the relationship between N and crop rotation is very important in N management decisions (42). Although there are several benefits to crop rotations (e.g., lower fertilizer N input because of residual N from legumes, erosion reduction, minimization of crop diseases, reduction in insect problems), relatively little attention has been given to impacts of NO_3^- N leaching by most crop sequences (32). Proper crop sequencing can sometimes be used to improve fertilizer-N efficiency and reduce nitrate leaching. Johnson et al. (43) showed that unfertilized soybeans scavenge large quantities of residual N fertilizer from a previous corn crop. Monocultures of crops, such as corn, winter

wheat, and sorghum, require high N input. It is possible to reduce N use in a farming system by rotating these crops with small grains, which require less N, or with legumes, which may require no added N. This type of crop diversification should result in less N in the soil profile and, in general, less leaching.

The combination of crops grown in a rotation also affects the potential for groundwater contamination by N. For example, less N moves beyond the rooting zone with a corn–soybean rotation compared to continuous corn (44). Including perennial legumes or non leguminous crops in rotations also decreases NO_3^- -N losses. Baker and Melvin (45) found much lower NO_3^- -N concentrations beneath alfalfa than for corn or soybean.

The key to successful erosion control with crop rotations is soil cover. Fallow land has the highest erosion potential in any cropping system. Crop rotations that include small grains or other cover crops create a vegetative cover, which protects the soil surface from erosion (46). Grasses and legumes or small grains provide more protection against water erosion than row crops (47). Cropping systems with a higher frequency of sod reduce soil erosion. Growing cover crops with low residue crops and rotating high residue crops with low residue crops are also effective erosion control practices. Some crop rotations do not reduce erosion unless other practices, such as cover crops and residue management are followed (48). Crop rotations that use the land more intensively, such as corn, wheat, and soybeans grown in 2 years, produce larger amounts of biomass during the rotation and are more effective in reducing erosion than a continuous cropping sequence (49).

Tillage Practices. Soil erosion and runoff contribute to the degradation of surface water quality. Tillage management is an important factor to be considered when attempting to reduce soil erosion and subsequent sedimentation (50). Tillage practices that leave the soil surface unprotected greatly increase the potential for runoff, erosion, and nutrient losses from fields (44). Conventional tillage buries the protective crop residue cover and disturbs the soil structure; increased rainfall may then lead to amplified runoff and erosion (51). Conservation tillage practices, which reduce runoff and erosion, reduce N losses (52–54). Such practices leave greater amounts of crop residue and cause less soil disturbance. Both factors help reduce runoff and erosion, reduce transport of soluble and sediment-attached chemicals, and promote infiltration (55). Baker and Lafen (56) concluded that conservation tillage should reduce soil erosion by 75–90% (depending on the amount of surface residue) compared to conventional tillage. Conservation tillage also increases infiltration and reduces average annual runoff volumes by about 25% compared to conventional tillage (57). The majority of the N in surface runoff is associated with sediments, so conservation tillage usually results in a net decrease in N losses.

Several of the factors that reduce surface runoff promote an increase in subsurface flow (32). Conventional tillage disrupts surface soil structures, destroying macropores in the tilled layer. It also enhances the mineralization

of organic N from crop residues and soils and increases the accumulation of residual NO_3^- N. In recent years, technological innovations have led to reduced levels of tillage in many areas. This trend seems to protect groundwater in some areas because of reduced N mineralization and increased N immobilization (44). The effect of tillage on macropores and the flow of water and chemicals through macropores are important (58–60). However, using conservation tillage, macropores remain intact from the soil surface to their full depth in the soil profile (61–63). Macropores potentially increase the amount of water entering the soil; this increase in infiltration leads to more leaching. Tillage has both advantages and disadvantages with respect to groundwater; information from studies on local soils should be used to determine if conservation tillage would have positive or negative impacts on subsurface water quality.

Fertilizer Management

Nitrogen Rate. The one most important practice affecting the N contribution to groundwater is the rate of N application (64). Excessive application of N fertilizer increases the likelihood that N is leached into surface water and groundwater (47). The amount of N from other sources should be given appropriate credit to avoid excess fertilizer N. The soil can supply N to plants as mineral N, usually NO_3^- N, and as N mineralized from organic matter. There can be substantial overwinter carryover of residual NO_3^- (65), which would allow reducing of subsequent N fertilizer applications.

Nitrogen rate recommendations also include credits for N fixed by legumes. Winter leguminous cover crops and legumes in rotation can provide a significant amount of N for a subsequent nonleguminous crop (66–68). Some legumes can provide as much as 120 kg N/ha, which shows strongly that credit needs to be given for legume N to reduce excessive N fertilizer applications (32). Similarly, N application rates can be significantly adjusted to account for manure application. By failing to account for these sources of N in determining the correct application rate, a surplus of N may be created; this surplus may leach to groundwater (69).

Similarly, N credits need to be given for manure applications. Recent reports of long-term manure applications indicate that high rates of application can pose a hazard for NO_3^- -N contamination in groundwater (32).

Fertilizer Placement. Correct fertilizer placement is a BMP that can help protect both surface and groundwater quality from nutrient pollution. The two most often used fertilizer placement strategies are banding and broadcast placement (70).

The efficiency of N fertilizer is generally increased by band placement that provides many agronomic and environmental advantages over broadcast applications (71). Banding causes an increase in plant N use efficiency. High crop-use efficiency leaves less N in the soil to leach into groundwater (70). Banding of N below or with the seed is by far the best N-BMP to protect surface water quality because no N is left on the soil surface for erosional loss. A broadcast-incorporated application leaves a portion of

the N fertilizer on or near the soil (70). A study in Montana found that spring wheat yields are approximately 6% higher when urea was banded 1 inch below the seed and between rows compared to broadcast (72).

Fertilizer Timing. The timing of N fertilizer application is an important factor that affects the efficiency of fertilizer N because the interval between application and crop uptake determines the length of exposure of fertilizer N to loss processes, such as leaching and denitrification. Timing N applications to reduce the chance of N losses through these processes can increase the efficiency of fertilizer N use (21).

Fertilizer applied in the fall causes groundwater degradation. Fertilizer-N applications should be managed very carefully on sandy or gravelly soils due to the high potential for leaching. Fall applications are not recommended on these types of soils (44). Fall applications of anhydrous NH_3 or urea on finer textured soils should be delayed until soil temperature reaches 45 °F or less when conversion to NO_3^- is slow, hence, the potential for NO_3^- -N leaching losses is reduced. Split application of N fertilizers as opposed to a single application can reduce potential N losses by up to 30% and thereby reduce groundwater contamination (73).

Organic Wastes. Animal manure is a good source of N. It benefits plant growth, improves soil structure, and provides nutrients. However, manure can become an environmental problem when it pollutes groundwater and surface water as a result of improper handling, storage, and application on erosion-prone soils (74). This is primarily due to the particulate nature of the material compared to the more soluble forms of N in inorganic fertilizers (44).

The other major concern with manure is the movement of NO_3^- N to groundwater. Manure contains mostly organic and NH_4^+ N. These forms of N do not move through soils. However, soil microorganisms can convert these compounds to soluble NO_3^- N that may leach to groundwater. Nitrogen not mineralized in the first year after application of manure generally becomes part of the soil organic matter and releases N relatively slowly. Constant annual manure applications that supply N to meet the entire crop demand ultimately cause excessive fertilization. Potential pollution problems resulted when manure was applied on a long-term basis at recommended rates (475 kg/N ha) in Alberta, Canada (75).

To use manures efficiently as a nutrient source for crop production, the spread pattern must be uniform throughout the field (76). Nonuniform applications of manure result in improper nutrient crediting and can increase the possibility of overfertilization, which threatens groundwater and surface water quality (77). Manure should be applied when crops have the highest need for nutrients. The quicker the nutrients are used, the less the chances of pollution due to runoff into surface watercourses or leaching into groundwater sources.

Manure should be stored in properly designed facilities that are protected from excessive runoff, flooding, or overflow that would allow contamination of surface

water. Injecting manure is especially important in preventing contamination of surface water supplies. Injection methods place liquid manure below the soil surface, thus eliminating surface runoff on sloping soils and volatilization of ammonia from the manure on all soils (76).

FUTURE DIRECTIONS

As Owens (32) points out, much is known about using N for efficient crop production and minimizing its adverse impacts on surface and subsurface water quality. Nevertheless, there are areas where further knowledge is needed to fill in the gaps and to improve upon what is already known. Following Owens (32), the areas mentioned here demand particular attention in research:

- *Subsurface water quality*: Much of the reported research in this area has been conducted with soil columns and lysimeters. There is need for more groundwater quality research on a field or watershed scale. A systems approach, in which existing knowledge is harnessed into quantitative, explanatory models, is more attractive and acceptable.
- *Urban use*: Fertilization of lawns and turf areas and the subsequent impacts on groundwater quality need more regulatory attention. In many states, there is no environmental law protecting water resources from overfertilization of privately owned lawn or gardens, although according to a Gallup poll, 22 million U.S. homeowners spent more than \$14.6 billion on professional landscape/lawn care/tree care services in 1997. Research on this aspect of N management has been limited and needs to be revisited as a potential major source of N pollution.
- *Soil N Availability Tests*: Crops are probably more deficient in N than in any other element, and yet there are no widely accepted methods of testing soils other than testing for residual NO_3^- N and NH_4^+ N (78). Considerable attention needs to be directed toward developing soil tests that provide an accurate in-season assessment of N need for a crop, yet take into account variation in soil types and kinds of crop while recommending fertilizer N needs.
- *Education*: Knowledge about N fertilizer efficiency and techniques for minimizing nitrate leaching are available but are not consistently used. The real value of continued research will be the adaptation and implementation of these technologies, which require a major educational/outreach effort.

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SOIL PHOSPHORUS AVAILABILITY AND ITS IMPACT ON SURFACE WATER QUALITY

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Phosphorus (P) is ubiquitous in soils and is an essential element for plant and animal growth. Without enough soil P, crop yields will be low. However, excess soil P may become a part of surface runoff and can accelerate eutrophication of receiving fresh waters. Eutrophication has been identified as the main problem in U.S. surface waters that has impaired water quality. Although P and nitrogen (N) both influence eutrophication, P is generally the limiting nutrient in freshwater bodies. The degree of P input in surface waters depends on the soil availability of P and the degree of transport of dissolved and particulate P, especially in agricultural lands that have been fertilized for a long period of time. The availability of P in soils is controlled by various factors, such as soil pH, clay content, soil organic matter, amorphous Fe/Al oxide and carbonate content, soil moisture content, and degree of soil compaction. The amount of available P is generally estimated by a variety of soil-test methods, including both chemical extractants and nondestructive techniques simulating plant root action. Phosphorus transport from the source to surface waters is controlled by the rate of erosion and surface runoff. Hence, reducing P loss in agricultural runoff may be brought about by implementing proper source and transport control strategies. Popular source control techniques include soil P management, estimating thresholds for environmental risk assessment, manure management, and refining feed rations, common transport management techniques include conservation practices such as reduced tillage, cover crops, and buffer strips in critical areas of P export from a watershed.

Phosphorus makes up about 0.12% of the earth's crust. It forms complex compounds with a wide variety of elements—about 150 minerals are known that contain at least 0.44% P (1). Phosphorus is ubiquitous in soils. It

is found naturally in all soils (1) and ranges between less than 0.04% in the sandy soils of the Atlantic and Gulf coastal plains to more than 0.3% in the soils of the northwestern United States (2). Phosphorus is an essential element for plant and animal growth. Its input has long been recognized as necessary to maintain profitable crop and livestock production (3). Without enough soil phosphorus, crop yields will be low (4). However, many soils in the United States reportedly contain P well in excess of crop needs, which may enter lakes and streams in runoff from landscapes that drain to surface water bodies. Sources of P contamination of surface waters are numerous and include both nonpoint and point sources, such as agriculture, municipal sewage treatment plants, individual septic treatment systems, decaying plant material, runoff from urban areas and construction sites, stream bank erosion, and wildlife (5). Drainage of P-rich runoff to surface water bodies may increase their biological productivity by accelerating eutrophication, the natural aging of lakes or streams brought on by nutrient enrichment. The United States Environmental Protection Agency (USEPA) has identified eutrophication as the main problem in U.S. surface waters that have impaired water quality (6). Because it causes increased growth of undesirable algae and aquatic weeds, as well as oxygen shortages, eutrophication restricts water use for fisheries, recreation, industry, and drinking (7). In addition, associated periodic surface blooms of cyanobacteria (blue-green algae) occur in eutrophic drinking water supplies and may pose a serious health hazard to livestock and humans. Although eutrophication is a natural process, it can be hastened by changes in the land use of a watershed that increase the amount of nutrients added to an aquatic system (7). Recent outbreaks of the dinoflagellate *Pfiesteria piscicida* in the eastern United States, most notably in the Chesapeake Bay, have been linked to high nutrient levels in affected waters (3). Nitrogen (N) and P both influence eutrophication, but P is the critical element in most freshwaters (7). Due to low natural levels of P, biological productivity in surface waters is usually limited by P availability (5). The identification of P as a major cause of eutrophication in surface waters led to early emphasis on point sources delivering P to surface waters, for example, sewage systems delivering P-enriched wastewater. According to the National Research Council (8), overall trends show about equal numbers of U.S. rivers that have increasing and decreasing P loads. In general, the decreases are linked to point-source reductions, and the increases are linked to nonpoint source increases that result from increased agricultural land use (9). This review will address the concept of soil availability of P in agricultural lands and the relationship between P availability, P transport, and surface water eutrophication. Popular practices for P management will also be discussed.

PHOSPHORUS IN SOILS

The total P content of most surface soils is low, averaging only 0.6%. This compares to an average soil content of

0.14% N and 0.83% potassium (K) (2). In most soils, the P content of surface horizons is greater than that of the subsoil. Except in special situations, added P tends to be fixed by the soils where it is applied and allows little movement down through the soil profile (7). In addition, P is cycled from roots to aboveground parts of the plant and redeposited in crop residues on the soil surface. This builds up organic material and stimulates biological activity in surface layers. In reduced tillage systems, fertilizers and manures are incorporated only to shallow depths, thereby exacerbating P buildup in the top 5 inches of soil (7).

Many factors influence the P content of soil. Among these are (1) the type of parent material from which the soil is derived, (2) the degree of weathering, and (3) climatic conditions. Among other factors influencing soil P levels are the rates of crop removal, erosion, and P fertilization (2).

There are two primary forms of P in soil systems—organic and inorganic. The organic form of P in many soils represents more than half of the total P. Organic P has three different sink/sources (10), namely nucleic acids (e.g., guanine, adenine, thymine, uracil), phosphate esters of inositol (e.g., phytin), and phospholipids (e.g., lecithin). Biological processes in the soil, such as microbial activity, tend to control the mineralization of organic P. Mineralization is the breakdown or conversion of organic P to inorganic solution P (7). The inorganic form of P in soils (orthophosphates) exist in various forms: (1) oxide–phosphate complexes, such as clay mineral edge-P complexes, and (2) minerals of apatite (Ca phosphates), Mn phosphates, Fe phosphates, Al phosphates, and to a lesser degree, pyrophosphate (10).

AVAILABILITY OF SOIL PHOSPHORUS

Loosely defined, P availability refers to the amount of soil P that is readily available for uptake by plants. If the amount of available P exceeds plant needs, the excess P may become available for drainage to surface water bodies as part of the surface runoff, thereby degrading surface water quality. As a rule of thumb, soluble P is considered the most available. However, the solubility of many inorganic P compounds, as well as organic P, is generally low, and only small amounts of soil P are in solution at any one time. In fact, most soils contain less than a 1 lb acre⁻¹ of soluble P (2). Soluble P in soils, either from fertilizer or natural weathering, reacts with clay, iron, and aluminum compounds in the soil and is converted to less mobile (hence, less “available”) forms by the process of P fixation. Because of the fixation processes (e.g., sorption and precipitation), P moves very little in most soils, stays close to its place of application, and crops seldom absorb more than 20% of fertilizer P during the first cropping system after application. This fixed, residual P remains in the rooting zone and becomes slowly available to succeeding crops (2).

Factors Controlling Availability of Soil Phosphorus

Soil P availability is influenced by the following factors (2):

- *Soil pH*: Soil pH affects both adsorption and precipitation of phosphorus. Precipitation of P as

slightly soluble Ca phosphates occurs in calcareous soils whose pH values are around 8.0. Under acidic conditions, P is precipitated as Fe or Al phosphates that have low solubility. From a precipitation standpoint, the maximum availability of P generally occurs in the pH range of 6.0 to 7.0 (2). As a typical oxyanion, phosphate tends to be adsorbed more at lower pH than at higher pH. Hence, P generally tends to be more available in high pH soils, unless precipitated as low-solubility basic salts. Amorphous Fe and Al oxides that have high specific surface strongly adsorb P in soil systems (11).

- *Soil Organic Matter*: High amounts of organic matter in soils promote mineralization of organic P, thereby providing available P for plant growth. Organic matter also acts as a chelating agent, thereby preventing the formation of insoluble iron phosphates. Application of organic materials such as manure, plant residues, or green manure to soils that have high pH values increases the availability potential of mineral forms of P in the soil (2).
- *Clay Content*: Fine-textured soils such as clay and clay loams have greater P-fixing capacity than coarse-textured sandy soils. Among soils of similar clay mineralogy, P fixation increases as clay content increases and crystallinity decreases (12). Among clay minerals, 1:1 type clays (kaolinite) have greater P-fixing capacity than 2:1 type clays (e.g., montmorillonite, illite). Soils formed under high rainfall and high temperatures generally contain large amounts of kaolinitic clays and therefore have relatively lower soil P availability.
- *Soil Aging*: Fixation of soil P increases with time of contact between soluble P and soil particles. Consequently, more efficient use of fertilizer P is generally obtained by applying the fertilizer shortly before planting the crop (2). As soil ages, the possibility that P binds to soil particles increases, thereby decreasing its potential availability.
- *Soil Compaction and Soil Moisture*: Soil compaction reduces the soils oxygen supply and decreases the ability of plant roots to absorb soil P. Compaction reduces aeration and pore space in the root zone. This reduces P uptake and plant growth. Compaction also decreases the soil volume that plant roots penetrate and limits their access to soil P. Many studies have been conducted on the effect of soil moisture on P availability and response to applied P (13). Power et al. (14) reported that 53% of the variation in the fertilizer response of spring wheat on medium P soils resulted from variation in soil moisture. MacKay and Barber (15) showed that P availability decreased at soil moisture contents above and below field capacity because of the effects on both P diffusion and root growth rates. Uptake of P by corn dropped more than total plant weight when soil moisture was decreased below field capacity, indicating that the decrease in uptake was not likely due only to decreased demand.

In addition to the factors discussed before, several other factors define soil P availability (13). These factors include,

but are not limited to soil solution P concentration, the amount of solid phase P that can enter the soil solution, the dissolution/desorption rate, the diffusion rate, Fe content, carbonate content, carbonate specific surface area, and organic P content (16–19).

Phosphorus Availability Indices

To evaluate P availability in soils, numerous soil tests have been developed that extract varying amounts of P, depending on the types of extractants used. The extractants can be generally classified into several categories (20):

- Water or unbuffered salt solutions (e.g., CaCl_2).
- Dilute concentrations of weak acids (e.g., lactate, acetate) with or without a complexing agent (e.g., F^- , EDTA).
- Dilute concentrations of strong acids (e.g., HCl , H_2SO_4) with or without a complexing agent (e.g., F^- , lactate, EDTA).
- Buffered alkaline solutions (e.g., NaHCO_3 , NH_4HCO_3) with or without a complexing agent (e.g., EDTA).
- Anion exchange resin or iron oxide impregnated filter paper strips.
- Isotopic exchange with ^{32}P .

Water derives readily soluble P; unbuffered/neutral salt solutions derive easily exchangeable P (21). Dilute strong acid solutions solubilize Ca-P, Al-P, and to a lesser extent, Fe-P (22). Complexing agents are generally introduced to prevent readsorption of P by Fe oxides (20). Anion exchange resins and iron oxide impregnated filter papers (nondestructive to soil components in contrast to chemical extractants) function as sinks that simulate the action of plant roots by continuously removing dissolved P from the soil solution. The quantity of P that is isotopically exchangeable within a specified time interval gives an estimate of labile (hence, available) soil P (20).

Criteria for a suitable soil test have been detailed in several reports (13). Thomas and Peaslee (23) felt that the extract should (1) remove a reproducible and consistent proportion of the soil's available P and (2) reflect the extent and nature of reactions between soil and added P. Barrow (24) also felt that a soil test was an integration of several factors affecting soil P status (e.g., soil P buffering capacity, P quantity, soil P reaction time) into one numerical value. Kamprath and Watson (25) noted that (1) the extractant should extract a proportionate fraction of available P from soils that differ in other properties, (2) the procedure should work with reasonable accuracy and speed, and (3) the soil test P value determined should be correlatable with P uptake. On the basis of the major soil types and their properties, various states in the United States have adopted different soil test methods to predict available P. A field rating system, the "P index," has been developed to assess the potential for soil P to contribute to surface water pollution (26).

TRANSPORT OF SOIL PHOSPHORUS

Phosphorus enters lakes and streams in runoff from landscapes that drain to those surface water bodies (5).

The eutrophication process is clearly accelerated by the addition of P to surface waters. All forms of P (soluble, adsorbed, precipitated, and organic) are susceptible to transport from source soils to water bodies. Transport of soil P occurs primarily through surface flow, although the background levels of P entering streams and lakes via subsurface flow certainly reflect the impacts of land use. For example, in soils that have considerable slope, surface or overland flow can infiltrate into the soil during movement down the slope, move laterally through a conduit in the soil, and reappear as surface flow (7). However, in most cases, the concentration of P in subsurface flow, it has been found is quite low and well below the eutrophication threshold. In overland flow (surface runoff), P may be dissolved in runoff water (soluble/dissolved P or DP). The small quantity of soluble or readily desorbable P in most soil environments is due to the low solubility of phosphate minerals and the considerable adsorption capacities of clays and Fe/Al oxides for P. These factors result in the majority of total P transport that occurs as particulate P (PP). Particulate P enters streams and other surface waters, first undergoing a solubilization reaction (e.g., desorption) before becoming available for aquatic biota. The mechanisms involved in soluble P transport include initial desorption or dissolution of P bound by soil particles, followed by water movement from the source soil to a stream or a lake that later intercepts a sensitive water body (27).

Factors Controlling Surface Transport of Soil Phosphorus

Two major factors influence P losses from the landscape and movement to surface water bodies (5):

- **Erosion:** Particulate P is the primary form of P contributing to degradation of surface water quality. Erosion largely determines PP movement in the landscape. Sources of PP in surface waters include eroding surface soil, stream banks, channel beds, and plant materials. The contribution of eroded soil material to P enrichment of surface waters is rather complex (5). First, part of the sediment eroded from the landscape deposits at the toe slope in depositional areas, along field boundaries, or in grassy riparian zones. Thus, sediment yield from the field to a water body is only a portion of that eroded from the slope. Second, during the movement of sediment, the finer fractions are preferentially transported, whereas larger particles tend to settle out. Therefore, the P concentration and reactivity of eroded particulate material is usually greater than that of the source soil. The enrichment of P may increase as much as six-fold as the relative movement of fine to coarser particles increases. For this reason, fine-textured soils (clay loam, silty clay loam, silty clay, or clay) have a higher potential of supplying P to surface waters if erosion occurs (5).
- **Runoff:** As discussed earlier, runoff water across the soil surface can contain significant concentrations of DP. In addition to the readily available P loss from surface soils via desorption/dissolution during rainfall, DP can also be lost from standing vegetation

via spring snowmelt because P contained in the tissue is released due to breakdown of plant cells by freezing and thawing. The removal of P from plant residue by overland drainage may account for differences among watersheds and seasonal fluctuations in P movement (5). Concentrations of DP in subsurface flow are low because the P-deficient subsoils sorb much of soluble P contained in the water percolating through the soil profile. Exceptions may occur in organic, highly permeable, peaty, or waterlogged soils, which tend to fix less P (7).

There is a well-defined relationship between the DP and PP fractions of total P in runoff as a function of soil erosion. As erosion increases, the PP fraction of total P increases, whereas the DP fraction decreases significantly (5). Irrigation, especially furrow irrigation, can significantly increase the potential for soil and water contact and therefore, can increase P loss by both surface runoff and erosion in return flows. Furrow irrigation exposes unprotected surface soil to the erosive effect of water movement. The process of irrigation also has the potential to increase greatly the land area that can serve as a possible source for P movement, a fact that is especially important in the western United States (3) where the surface water bodies suffer from an increased threat of eutrophication.

ENVIRONMENTAL MANAGEMENT OF SOIL PHOSPHORUS

When looking at management and/or remedial practices to minimize the environmental impact of P, several important factors must be considered. To cause an environmental problem, there must be a source of P (i.e., high soil P levels, manure or fertilizer applications, etc.), and it must be transported to a sensitive location (i.e., by runoff, erosion, or to a lesser extent, leaching). Hence, reducing P loss in surface runoff (thereby lowering the eutrophication potential of surface waters) may be brought about by source and transport control strategies (3).

Source Management

Source management attempts to minimize soil P buildup by controlling the amount of P applied to a given area. One of the most highly recommended avenues for source management is by measuring (and monitoring) soil test P in agricultural lands and estimating threshold levels for environmental risk assessment.

Decades of fertilization at rates exceeding those of crop removal have resulted in widespread increases in the P status of many agricultural soils, even to levels considered "excessive." Environmental concern has forced many states to consider developing recommendations for P applications and watershed management based on the potential for P loss in agricultural runoff (3). As a result, well-defined "agronomic" and "environmental" thresholds for soil test P have been developed for many states. The environmental threshold levels range from two times in Michigan (40 ppm P-agronomic and 75 ppm P-environmental) to four times in Texas (44 ppm vs. 200 ppm). However, one still cannot accurately project that a soil test level above an expected crop response level

exceeds crop needs and therefore is potentially polluting. What will be crucial in terms of managing source P based in part on soil test levels will be the interval between the critical soil P value for crop yield and runoff P (3).

Among the other popular source management techniques currently being implemented in farmlands are manure management (e.g., composting, lime or alum amendment, separation of solids from liquids); reduction of off-farm inputs of P in animal feeds (3); and controlling the rate, method and time of P application (5).

Transport Management

Phosphorus loss via erosion and runoff may be reduced by increasing residue cover on the soil surface through conservation tillage (5). Results from a flat, tiled, clay loam site in Michigan show significantly lower losses of sediment, total P, and soluble P for chisel tillage compared to moldboard plow tillage. However, sediment reductions were much greater than the reductions in total or soluble P. In other research, DP concentration in the runoff from no-till practices has often been greater than that from conventional practices (5).

Strategically placed and properly designed filter strips, it has been shown, effectively reduce erosion and P movement, especially PP (5). However, the strips have had little consistent effect on reducing DP concentrations.

Other common practices for reducing potential P transport include terracing, contour tillage, riparian zones, cover crops, and impoundments (settling basins). Basically, all these methods reduce the impact of rainfall on the soil surface, reduce runoff volume and velocity, and increase soil resistance to erosion. Most of these practices are more efficient in reducing PP than DP, and none of them should be relied on as the primary or sole practice to reduce P losses in surface runoff (3). Several researchers report little decrease in lake productivity (i.e., eutrophic plant growth) with reduced P inputs, following implementation of conservation measures (28,29). Thus, the effect of remedial measures in the contributing watershed will be slow for many cases of poor water quality (7).

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SOIL WATER ISSUES

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Landscape plants and *turfgrass* can eliminate erosion and conserve water most effectively by considering many variables in the makeup of the landscape. These variables include the sizes of mineral components of soil, the components of the root zone, *soil moisture capacity*, soil texture, soil pH, the soil's *biodegradable capacity*, and its mulching capacity. All these factors are explained in this entry.

Because water and soil are the lifeblood of the planet, any policy that governs water use or proposes water conservation must incorporate the dynamic relationship that takes place between plants, the soil and water. The soil on which all plants grow is a highly valuable natural resource. We have a responsibility to protect it from erosion and to conserve and enrich it for future generations, just as we have a responsibility to use our supplies of water judiciously and maintain or improve its quality. *Landscape* plants generally and *turf-grasses* specifically can help achieve both of these goals.

To realize these benefits, it's helpful to look at what scientists have learned about the characteristics of soil such as texture, size of soil particles, the presence of organisms, capacity to hold moisture, acidity–alkalinity, and the presence of pollutants.

Soils are not homogenous, inert materials. They are composed of mineral particles that include sand, silt, and clay, as well as living and dead *micro/macroflora* and fauna, chemicals, air, and water in various percentages.

SIZES OF THE MINERAL COMPONENTS OF SOIL

From largest to smallest, mineral components of soil consist of the following:

- Stones = 10 to 100 millimeters in diameter (25 per inch).
- Gravel = 2 to 10 millimeters
- Coarse sand = 0.2 to 2 millimeters
- Fine sand = 0.02 to 0.2 millimeter
- Silt = 0.002 to 0.02 millimeters

- Clay = smaller than 0.002 millimeter. Clay particles are so small that they are measured in microns (0.002 millimeter is equivalent to 2 microns)

In terms of comparative size, if we enlarge a clay particle to the size of an apple, then a silt particle on the same scale would be the size of a limousine and a medium sand particle would be the size of the White House in Washington D.C. Because soil particles have relative sizes this small, there are many of them. For example, a pound of medium sand contains about 2.5 million particles, a pound of silt contains more than 2.5 billion particles, and a pound of clay contains more than 40 trillion particles. On the basis of total particle surface, one pound of sand would account for 20 square feet; silt would present 220 square feet of surface, and clay would have 5500 square feet of particle surface (Fig. 1).

THE MULTIPLE COMPONENTS IN A LANDSCAPE ROOT ZONE

Also present in most soils are high microbe counts within the root zone. Often there are more than 900 billion for each pound of soil. In each 1000-square-foot surface to a 6-inch root-zone depth, there will be a total of about 45 quadrillion organisms. As these organisms complete their life cycle and die, they deposit into the soil some 10 pounds of nitrogen, 5 pounds of phosphorus, 2 pounds of potassium, a half-pound of calcium, a half-pound of manganese, and one-third of a pound of sulfur for each 100 pounds of dead organisms on a dry-weight basis.

Soil microbiological processes also convert organic matter into humus. This is an ongoing reaction of great importance. Humus helps to form and stabilize soil aggregates that are essential for deep and extensive root growth. Humus also contributes to the process within the soil that holds and releases nutrients for plant growth.

In addition, many small animals known as soil fauna occupy the root zones of plants and contribute to the living nature of the soil. Depending upon soil conditions that are favorable for these macroorganisms, from 1 million to 2 million may be present for each 1000 square feet of root zone. The live weight of these organisms would range from 15 pounds to 30 pounds per 1000 square feet.

Of course, the water molecule associated with the soil is exceptionally small. One fluid ounce of water contains approximately 1,000,000,000,000,000,000,000 (24 zeros, or one trillion trillion) molecules.

GRASS PLANTS HAVE A TREMENDOUS POTENTIAL FOR ROOT GROWTH

Grasses fit right in with the sizes and numbers of soil particles found within this fascinating system. For example, there may be as many as 35 million individual grass plants per acre, or about 800,000 per 1000 square feet. No other type of plant culture involves such crowding. Roots grow down into the soil, and it is there that grass plants have a tremendous potential for root growth—up to 375 miles of roots from one plant and as many as 14 million



Figure 1. In terms of comparative size, if we enlarge a clay particle to the size of an apple, then a silt particle on the same scale would be the size of a limousine, and a medium sand particle would be the size of the White House in Washington, DC.

individual roots that may have a total surface area of 2500 square feet. Thus, root numbers and surfaces fit well within the very small spaces surrounding aggregated soil particles.

It is important to understand soil properties so we can appreciate how turfgrass needs moisture to grow and enhance the environment (Fig. 2).

SOIL'S CAPACITY TO HOLD MOISTURE

Soils differ in their capacity to hold moisture. Heavier clay and silt soils hold more moisture. Sandy soils can lose moisture through leaching as it runs through the root zone and down into the subsoil. Grasses with well-developed,

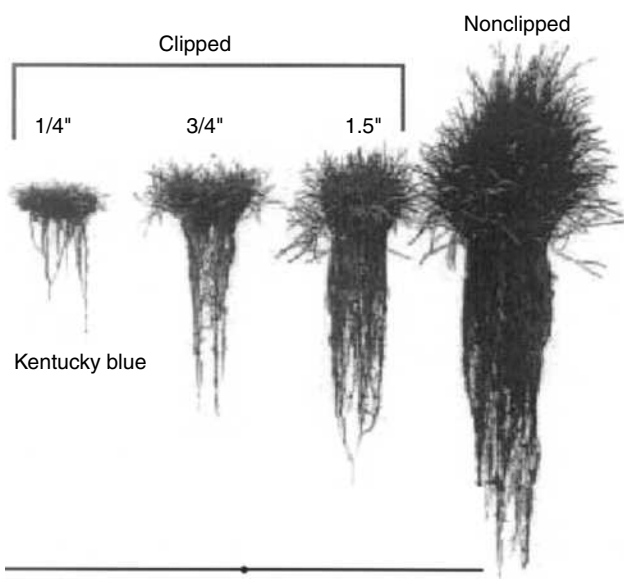


Figure 2. Higher mowing of turfgrass promotes a good root system. For Kentucky bluegrass, 1.5 inches is about right.

deep root systems add sufficient organic matter to help hold moisture in the soil and thus prevent leaching.

THE INFLUENCE OF SOIL TEXTURE ON WATER PENETRATION

The texture of the soil (as determined by the amount of sand, silt and clay) and the amount of thatch (organic deposit between green leaves and roots) influence the speed of water penetration into the soil. In general, heavy soils have many smaller pore spaces and take water in slowly. Sandy soils with fewer but larger pore spaces take moisture more rapidly, unless they are inherently hydrophobic or hard to wet. Soils and thatch that are hard to wet must be watered slowly with small amounts of water applied over longer periods of time to prevent runoff. Sandy soils require less water to penetrate to a given depth. Loam soils need intermediate amounts of water, and clay soils require more water to reach the same depth. As solid particles in the soil decrease on a percentage basis, moisture-holding capacity increases, and soil aeration decreases.

THE DEGREE OF ACIDITY AND ALKALINITY IN SOILS

Soil may either be acid, neutral, or alkaline. Soil pH (the measurement of the degree of acidity and alkalinity) is influenced by soil properties, biological influences, and climatic influences. Under acidic soil conditions, silt and clay particles tend to exist as individual units. Under more alkaline soil conditions, where calcium and magnesium are more plentiful, clay and silt particles group together to form granules. These provide improved soil structure, which results in more favorable balances of air and water in the soil. Where soils are acid and have poor structure, water penetration is much slower.

Soils become acid as carbon dioxide changes to carbonic acid in the soil, acid-reacting fertilizers are used on a continuing basis, or acid rainfall lowers soil pH values. Often a combination of all three of these causes occurs.

In addition to the effect of acidity on physical soil properties, nutrient fixation and availability also are modified, depending on the degree of acidity or alkalinity. For example, phosphates are most available from pH 5.5 to 7.5. Above and below these levels, phosphates are tied up with other minerals, and their availability is reduced. Regular soil tests can determine the need for lime or sulfur or for fertilizer mineral nutrients for specific plant types.

SOIL AS A BIODEGRADABLE AGENT

Biologically healthy soil is the best known medium for decomposing of all sorts of organic compounds, including pesticides and pollutants transported by air and water. These chemicals are known to be biodegradable. This is an ongoing process, which changes these substances into harmless compounds plus carbon dioxide and water. Limited prescribed use of pesticides is not harmful to beneficial soil organisms and should continue to be an important, well-accepted part of plant culture.

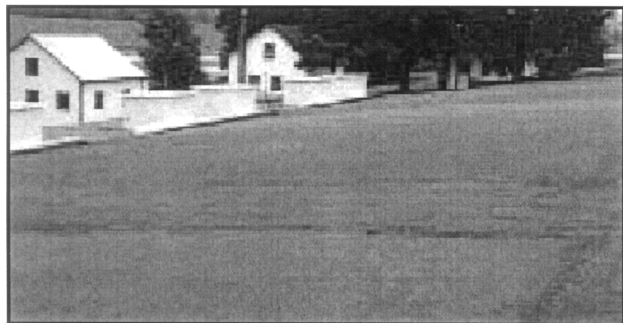


Figure 3. Dr. Thomas Watschke, Pennsylvania State University, created this highly controlled watershed site documenting that established turfgrass has a dramatic, positive effect on reducing nutrient and pesticide pollutants from water runoff.

Water and any pollutants associated with it infiltrate into the ground more quickly on grass-covered soils than on any other surface. Thus, runoff is diminished. Infiltration rates may be as high as 7 inches an hour on sandy soils and as low as 0.10 inch per hour on clay soils. Thus, recharge of purified groundwater is an important benefit. An acre left in open space provides an average of 600,000 gallons of recharge each year in humid regions. Grasses may use up to 10% of the water infiltrated, leaving 90% to recharge the local aquifer (Fig. 3).

GRASS GROUNDCOVER PROVIDES A LIVING MULCH

Good horticultural practice involves using mulches to conserve soil moisture and increase soil productivity. Unlike many landscape plants that are either widely spaced or simply annual in their growth habits, grass groundcover provides a living mulch over the soil surface. This is essentially perennial and provides long lasting soil and water conservation benefits.

Instead of viewing green-lawn groundcovers as static liabilities, these areas can be seen as dynamic, ever-changing populations of plants and animals living within and above the soil. All grasses are natural soil builders. Particularly in residential areas, lawns and landscapes help sustain the soil. Within the soil are large populations of micro- and macroorganisms that are highly competitive. These create a living, moist soil environment best suited to sustaining productive landscape soils, while at the same time purifying our water supply.

WATER SPREADING

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INTRODUCTION

In arid and semiarid regions, irrigation water shortage forces farmers to use floodwater for crop production. Water

spreading is a form of runoff farming. It is a form of flood irrigation accomplished by diverting and spreading runoff from sloping areas over adjacent flood plains (1). Water spreading involves the construction of basins, pits, or barriers in or near natural stream channels to impound water and cause it to infiltrate the ground surface rather than leave the basin as surface runoff (2,3).

In other words, diverting or collecting runoff from natural channels, gullies, or streams with a system of dams, dikes, ditches, or other means, and spreading it over relatively flat areas is called water or floodwater spreading (1,4,5).

In a large part of the world, runoff water harvesting is best known and practiced in the semiarid areas (6,7). Runoff farming, which is identical with water harvesting for irrigation purposes, was already used for more than 2000 years. For example, runoff irrigation facilities in the Negev Desert in Israel were built 1300 to 2900 years ago (8). Flood irrigation also has been practiced in other parts of the world, such as Iran, for many years. Traditional flood irrigated farms, called band-sar, can be observed in 400,000 ha of Khorasan, the northeast province of Iran. Band-sar controls about one-fiftieth of 200–250 mm annual precipitation and one-fourteenth of 14,000 MCM annual surface runoff of Khorasan Province, Iran (9). Many sources exist addressing the floodwater spreading affects on soil, groundwater, wild habitats, etc. Baghernejad (10) studied physicochemical and micromorphological changes in topsoil because of flood spreading in Damghan Playa, Iran. Kowsar and Yazdian (11) investigated the fate of the dissolved nitrogen in floodwater that reached the watertable in Bisheh Zard Basin, Iran, because of flood spread over the land. The floodwater was from a geological formation containing NO_3 and NH_4 . They found that the NO_3 exceeded the N required by plants of the region (rangelands and forest area), and purification of water must be in order to prevent the groundwater pollution. Nejabat (12) reported that a 2500 ha floodwater spreading system affected about 7500 ha of adjacent area in Gareh Bygon, Iran. He used a technique on the base satellite imagery to determine the increase in cultivated and rangeland areas and the decrease in wind erosion in water spread area.

Floodwater spreading has several advantages that are summarized by FAO (13) as follows:

- a. Controlling floodwater damage
- b. Reducing erosion and sedimentation
- c. Increasing groundwater recharge
- d. Improving plant cover and vegetation of the land
- e. Increasing economic return for people
- f. Increasing rangeland productivity
- g. Increasing animal and agricultural production
- h. Increasing soil fertility
- i. Reclamation of renewable resources
- j. Land use change by providing an optimized model for use of flow efficiency land and farming
- k. Job creation by fruit trees plantation and increasing the income of local people

- l. Improving the ecological balance of the environment
- m. Reclamation of the pasture and vegetation cover
- n. Providing ground for public participation
- o. Providing ground for sustainable development

Floodwater spreading also has some disadvantages (14), listed as follows:

- a. Affected by water shortage and climatic risks
- b. Affected by farmers' experiences rather than on the scientifically well-established techniques
- c. The possible conflicts between upstream and downstream users
- d. Possible harm to fauna and flora adapted to running waters and wetlands
- e. Requirement of relatively large labor inputs
- f. Need to a relatively large natural watershed area
- g. Need to appropriate investment and policy reforms to enhance the contribution of rainfed agriculture

In this article, water spreading as an irrigation technique will be introduced.

FLOODWATER SPREADING

Water harvesting refers to methods used to collect water from sources where the water is widely dispersed and quickly changes location or form and becomes unavailable or that is occurring in quantities and at a location where it is unusable unless some intervention is practiced to gather the water to locations where it can provide benefits (15). A number of water harvesting techniques exist that are practiced in many of the water scarce areas of the world, including (15,16):

- a. Roof rainwater collection
- b. Terracing
- c. Small dams
- d. Runoff enhancement
- e. Runoff collection
- f. Water holes and ponds
- g. Tanks
- h. Floodwater spreading

Floodwater spreading or spate irrigation is one of the common methods of runoff farming in arid and semiarid areas with rainfall intensity more than the infiltration rate of the catchment. USDA emphasized that a major difference between the water spreading system and other surface irrigation techniques, such as border and basin systems, is that the water spreading system is designed to meet precipitation and runoff conditions of the area and apply runoff to cropped fields, whereas other systems are designed to deliver water in accordance with plant needs (17).

Floodwater spreading has at least an inlet to divert flow from natural channels to the system, a delivery channel, and some contour spreader channels to spread water on

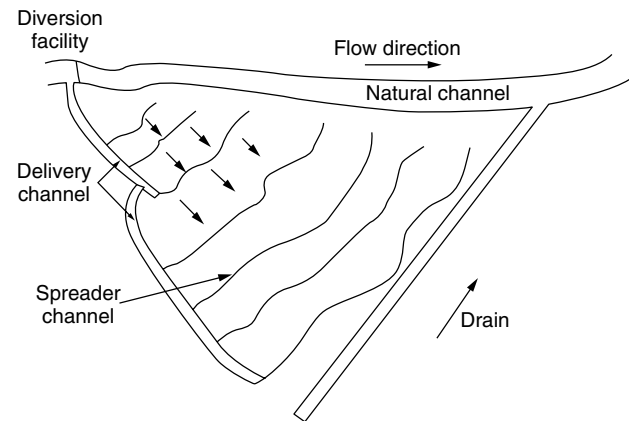


Figure 1. Schematic layout of a floodwater spreading system.



Photo by: R. Hasanzadeh

Figure 2. Diversion dam of Chikhab flood spreading system, Dehloran, Iran.



Photo by: R. Hasanzadeh

Figure 3. Intake and sediment control facilities of Poldasht flood spreading system, Makoo, Iran.

the ground as overland flow. Figure 1 shows a schematic layout of a floodwater spreading system. Figures 2, 3, 4, 5 show a diversion dam and intake system with sediment control facility, a delivery channel, and the spreaders of flood spreading system, respectively.

After a flood is diverted from the natural waterway, it reaches the spreading fields by the diversion or delivery channel and contour spreader. Sheet flow over



Photo by: R. Hasanzadeh

Figure 4. Diversion or delivery earthen canal of Sarchahan flood spreading system, Hajiabad, Iran.

Photo by: R. Hasanzadeh

Figure 7. A cemented sandbag-type spillway of Poldasht flood spreading, Makoo, Iran.

Photo by: R. Hasanzadeh

Figure 5. Contour spreader and downstream cultivated field of Poldasht flood spreading system, Makoo, Iran.

Photo by: R. Hasanzadeh

Figure 8. A gabion-type spillway of Chikhab flood spreading, Dehloran, Iran.

the spreading area provides sufficient time to infiltrate more and more water to the soil and completes water holding capacity of the plant root zone. Excess water will be drained to a downstream contour spreader from one or more spillways. Figure 6 shows a schematic flood spreading system with some spillway, and Figs. 7, 8, 9 show three types of spillways, a cemented sandbag, a gabion type, and a cemented rockwork spillway, respectively.

GOVERNING EQUATIONS

Overland flow because of floodwater spreading is on the base of shallow water hydraulics. The general characteristic of shallow water flow (SWF) is that the

vertical scale is much smaller than typical horizontal scale, which is true in many situations. SWF surface is nearly horizontal, which allows a considerable simplification in the mathematical formulation and numerical solution by assuming the pressure distribution to be hydrostatic. However, they may not exactly be two-dimensional (2D). The flow exhibits a 3D structure, because of bed friction, just as in boundary layers. Moreover, density stratification, because of a difference in temperature or salinity, causes variations in the third direction. In many SWFs, these 3D effects are negligible, and it is sufficient to consider the depth averaged form, which is 2D in the horizontal plane (18). This restricted form is commonly indicated by the term shallow water equations (SWEs).

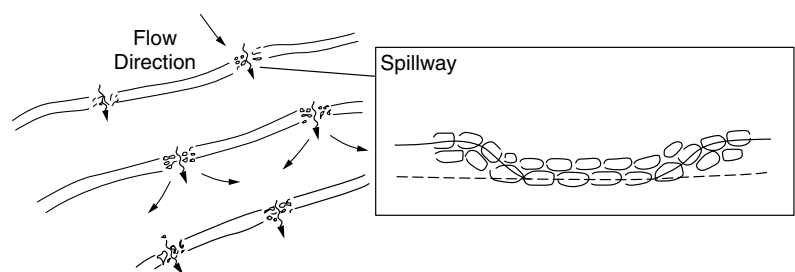
**Figure 6.** Schematic spillway (16).



Photo by: R. Hasanzadeh

Figure 9. A cemented rock work spillway of Chikhab flood spreading, Dehloran, Iran.

In this article, we limit our attention to incompressible fluid (water) and constant temperature and salinity. By assuming hydrostatic pressure, and with no Coriolis and wind effects and negligible effective stresses, the 2D depth averaged equations of motion are written as (19,20):

Continuity equation:

$$\frac{\partial h}{\partial t} + \frac{\partial(uh)}{\partial x} + \frac{\partial(vh)}{\partial y} = -I \quad (1)$$

x-momentum equation:

$$\begin{aligned} \frac{\partial(uh)}{\partial t} + \frac{\partial(u^2h)}{\partial x} + \frac{\partial(uvh)}{\partial y} + g \frac{\partial}{\partial x} \left(\frac{h^2}{2} \right) \\ = gh(S_{0x} - S_{fx}) + \frac{uI}{2} \end{aligned} \quad (2)$$

y-momentum equation:

$$\begin{aligned} \frac{\partial(vh)}{\partial t} + \frac{\partial(v^2h)}{\partial y} + \frac{\partial(uvh)}{\partial x} + g \frac{\partial}{\partial y} \left(\frac{h^2}{2} \right) \\ = gh(S_{0y} - S_{fy}) + \frac{vI}{2} \end{aligned} \quad (3)$$

where h is flow depth; u and v are flow velocities in x and y directions, respectively; I is the infiltration rate; t is time; g is gravitational acceleration; S_{0x} and S_{0y} are ground surface slopes because of x and y directions, respectively; and S_{fx} and S_{fy} are friction slopes in the direction of x and y axes, respectively.

Numerical solutions of SWEs with full or simple forms were described in several articles. Menendez (19), Akanbi and Katapodes (20), and Vreugdenhil (18) introduced numerical methods of solution of 2D SWEs. Arasteh and Vahhadj (21) developed a quasi2D finite difference semianalytical model of kinematic waveform of SWEs for Bisheh Zard flood spreading system, Iran. Arasteh (22) used an implicit finite difference scheme for the mentioned quasi2D model. Arasteh and Banihashemi (23–25) developed a solution method for implicit four points finite difference method to solve 1D kinematic wave SWE, called

growing coefficient matrix, and used it for a typical floodwater spreading plot in advance phase of flow to the end of the spreading area.

DESIGN

NRCS (4) introduced the main conditions for applying the floodwater spreading as:

- Soils have suitable intake rates and adequate water holding capacities for the crop to grow
- Soils are suitable for production of feed, forage, or grain crops
- The topography is suitable for the diversion or collection and spreading of water to achieve the desired results
- Runoff or stream flow is available at the time of the year and in a volume sufficient to increase plant growth
- Flows can be collected or delivered and spread and exceed water returned without causing excessive erosion
- Fish and wildlife will not be significantly adversely affected
- Grazing of the spreading area can be controlled

NRCS (4) also described the water quantity and quality requirement of flood spreading. FAO pointed out that water spreading can be used under 100 to 350 mm annually with a slope of less than 1% and even topography (16). USDA divided floodwater spreading systems into flow-type and detention-type systems (17). The flow-type systems incorporate free drainage from the irrigated area and are divided into spreader-ditch, syrup-pan, and dike-bleeder types. Detention-type systems retain the applied water on the irrigated area until it has infiltrated and are divided into manual and automatic inlet control subsystems. Apart from the type of system, two parameters must be determined as design criteria of the system, design application depth (DAD) and divertible volume of flow rate (DV or DQ). USDA stated that DAD could be determined by a relationship between intake rate, flow duration, or time of concentration, and DV or DQ could be determined as a ratio of volume of storm or peak discharge of upstream catchment.

SUITABLE SITES

<http://www.fao.org/docrep/u3160E/33160e00.htm>
<http://www.fao.org/ag/agl/aglw/wharv.htm>
<http://www.eng.warwick.ac.uk/ircsa/index.htm>

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SPRINKLER IRRIGATION

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In arid and semiarid environments, supplemental irrigation has become a prerequisite to minimize risks of crop failures in both rain-fed and irrigated agricultural areas. Usually, surface irrigation, in which the entire or most crop area is flooded, is used for irrigating these croplands (Fig. 1). More often, however, water management skills on the farm are lacking, water supplies may be uncertain, and so irrigation efficiency tends to be below.

Because of the present scarce global water resources, improving irrigation efficiency—getting more crop per drop—and making the best use of water for agriculture are prerequisites for the future (1). Shifting from surface irrigation to sprinkler irrigation, which is a method of irrigation under pressure in which water is sprinkled



Figure 1. Surface irrigation.

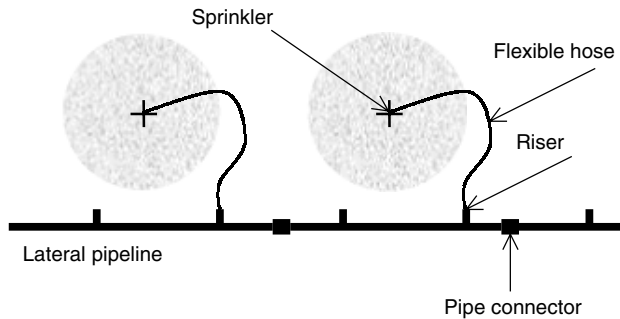


Figure 2. Lateral pipeline.

to imitate rainfall, will contribute substantially to these goals. In contrast to surface irrigation, sprinkler irrigation can help to irrigate frequently with just the right amount of water to avoid crop distress and to avoid overirrigation (Figs. 2 and 3).

In recent decades, various sprinkler irrigation methods and installations have been developed to meet the needs of farmers, particularly smallholders. Hand-move sprinkler irrigation is the most widely adopted and least expensive system operated with pressure ranges from 2.0–3.5 bars. The sprinklers are mounted on lateral pipelines temporarily laid across a field. To reduce labor requirements, this system was modified to hose-move sprinkler irrigation (Fig. 4). In this modified system, lateral pipelines are placed permanently, and the sprinklers, mounted on tripod stands, are not fitted directly to the lateral pipelines but are connected to them via flexible hoses. The hoses and sprinklers can be moved laterally on either side to cover a number of lateral positions (2).

Generally, in sprinkler irrigation, the uniformity of distribution is more sensitive to wind. However, this irrigation provides ‘air-conditioning’ effects on the crops if used in anti frost sprinkling and can also reduce pesticide applications if used as a spray. It is also fairly well established that a plan of heavy sprinkling at infrequent intervals will ordinarily produce a deeper root structure and thus better crop growth and yields than frequent, light sprinkling. However, light irrigation



Figure 3. Hand-move sprinkler irrigation.



Figure 4. Hose-move sprinkler irrigation.

may be beneficial when there is a minimum crop water requirement. Under shallow water table conditions, when groundwater contributes significantly to meet the crop water requirement, light irrigation is also required to dilute the salinity in the effective root zone. Sprinkler irrigation may be used to germinate and establish salt-sensitive crops by keeping the seedbed adequately moist and salt-free by uniformly applying small depths of irrigation water.

When sprinkler irrigation is done with saline water, it brings water into contact with foliage, and salt deposits on leaves may adversely affect some crops. However, the susceptibility of plants to leaf injury from saline sprinkled water depends on leaf characteristics that affect the rate of absorption and is not generally correlated with tolerance to soil salinity. The degree of spray injury varies with weather conditions, especially the water deficit of the atmosphere. Visible symptoms may appear suddenly following irrigation when the weather is hot and dry. Increased frequency of sprinkling, in addition to increased temperature and evaporation, increases the salt concentration in the leaves due to adsorption and results in leaf damage. However, irrigation at night (or any other low evaporation period) minimizes the salt concentration in the leaves due to adsorption (3). Furthermore, evaporation losses during sprinkling can also be minimized by avoiding midday irrigation.

High capital costs, nonavailability of needed equipment in the local market, energy dependence, and lack of local expertise to operate and maintain sophisticated hydraulic equipment are the main constraints that hinder the acceptability of the system by the farmers, particularly smallholders. For the same reasons, surface irrigation is still likely to be dominant in the future, even though it wastes water and is a major cause of waterlogging and salinization.

However, the integration of surface irrigation and sprinkler irrigation could help in controlling salinity in the root zone with minimum application of irrigation water by improving leaching efficiency. In this practice, sprinkler irrigation helps in uniform distribution of groundwater, and surface wetness due to sprinkling that helps in minimizing the application of surface water still can meet the crop water requirement. This surface wetness also

allows salts to dissolve in the root zone, which improves the leaching efficiency of surface irrigation. Therefore, the value of sprinkler irrigation can still be promised by promoting it together with surface irrigation (4).

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STOMATES

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Stomates or stomata (singular: stoma or stoma) are structures on the surfaces of plant leaves that allow gas exchange (transpiration, i.e., loss of water vapor, uptake of CO₂, and emission or uptake of O₂) between the interior of a leaf and the atmosphere.

Plant leaves are covered by a waxy layer, the cuticle, that prevents water loss from the cells on the leaf surface, or epidermis. The cuticle does not completely cover the leaves, however; it is interrupted by microscopic pores, surrounded by pairs of specialized guard cells (Fig. 1a). A stoma is a unit composed of a pore and its guard cells. Guard cells, unlike regular epidermal cells, usually contain chloroplasts (photosynthesizing organelles containing chlorophyll). In some plants, there are also additional, specialized subsidiary cells that differ in shape from regular epidermal cells and participate in the osmotic changes that drive guard cell movement.

Stomata can be found in all aboveground parts of plants but are more frequent in leaves. They usually appear on one (the lower) side of leaves but may appear on both. When stomata occur on the lower (abaxial) leaf surface, the plant is said to be hypostomatous; hyperstomatous plants have stomata only on the upper (adaxial) leaf surface, and plants with stomata on both surfaces are called amphistomatous. In this latter case, abaxial stomata are generally more numerous. The number of stomata per unit area is called the stomatal density. This figure may vary when plants are grown under different environmental conditions such as atmospheric CO₂ concentration.

The size and density of stomata vary widely among species (Table 1). When fully open, stomatal pores are

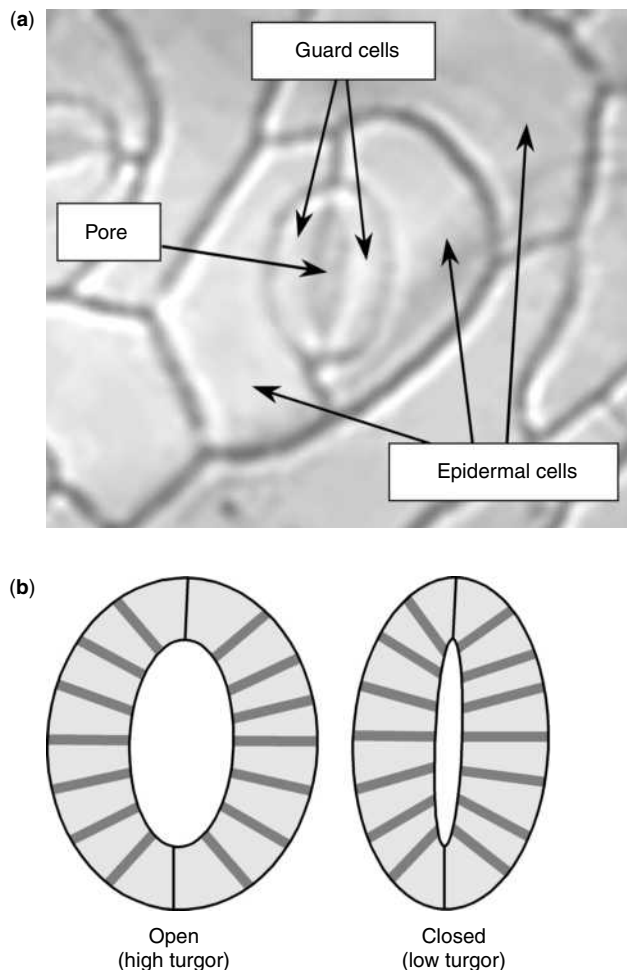


Figure 1. (a) Micrograph of a peanut (*Arachis hypogaea* L.) epidermis, showing the guard cells, the stomatal pore, and epidermal cells; (b) Diagram of open and closed stomata, showing cellulose microfibrils.

3–12 μm wide and from 10 to more than 30 μm long. Species with smaller stomata tend to have a higher density that can range from 60–80 per mm^2 in corn to 150 in legumes such as alfalfa and clover, 300 in apple, and more than 1000 in scarlet oak (1). Dicotyledon leaves with a netted venation pattern generally have a scattered arrangement of stomata, whereas monocotyledons and conifers tend to have their stomata arranged parallel to the main axis of leaves.

Some plants locate their stomata at the bottom of recessed hollows and sometimes grow hair-like structures above them, apparently to limit water loss by creating a high-humidity environment above the pore. Most plant leaves have small open spaces under the pore, called substomatal cavities (Fig. 2).

STOMATAL CONDUCTANCE MODEL

Transpiration and its dependence on stomatal control have been traditionally described by using an electric analog model based on conductances or their reciprocal, resistances. The basic assumption of the conductance

Table 1. Occurrence and Dimensions of Stomata for Several Representative Species^a

Plant Type	Representative Species	Common Name	Stomatal Density, Pores/mm ²		Guard Cell Dimensions, μm , for each Leaf Surface				Dimensions of Stomatal Pore, Abaxial, μm	
					Adaxial		Abaxial		Length	Depth
			Upper	Lower	Length	Width	Length	Width		
Moss	<i>Polytrichum commune</i>	Hair cap moss	16	16	46	15	46	15	15	12
Fern	<i>Osmunda regalis</i>	Royal fern	0	67			56	19	30	15
Conifer	<i>Pinus silvestris</i>	Scotch pine	120	120	28	7	28	7	20	6
Dicot tree	<i>Tilia europea</i>	European linden	0	370			25	9	10	8
Dicot herb	<i>Helianthus annuus</i>	Sunflower	120	175	35	13	32	14	17	
Dicot herb (xerophyte)	<i>Sedum spectabilis</i>	Autumn joy	35	56	32	10	33	10	20	18
Monocot herb	<i>Allium cepa</i>	Onion	175	175	42	19	42	19	24	18
Monocot C ₃ grass	<i>Avena sativa</i>	Oats	50	45	52	15	56	13	20	10
Monocot C ₄ grass	<i>Zea mays</i>	Corn	98	108	38	10	43	12	20	10

^aAdapted from Reference 1.

model is that the diffusion of gases within plant tissues and in the atmospheric boundary layer is restricted by resistances and that the resulting system of gas fluxes into and out of leaves can be described by the equations developed for analyzing electrical circuits (Fig. 3).

Nobel (2) described the diffusion of gases along stomatal pores by applying Fick's law and a stomatal conductance (which we will denote as g_w^{st}). This is a widely used parameter in modeling applications; stomatal conductance, the rate of flow per unit area of the leaf, is considerably easier to measure than the flux density within a stomatal pore. Stomatal conductance is closely related to stomatal aperture; Jones (3) showed how to estimate conductance from observed stomatal dimensions. Given the dynamic behavior of stomatal aperture, however, conductance is highly variable, especially in natural conditions. Time-averaged values are typically used in calculations.

The cuticular conductance, g_w^{cut} , is probably the most uncertain of all conductances in this model. Traditionally it has been assumed to be negligible, but experimental data show that cuticular transpiration varies greatly among species. Larcher (4) generalized that it constitutes 10–33% of total transpiration, but some researchers presented even higher values (5).

Nobel proposed calculating g_w^{ias} , the conductance of intercellular air spaces, based on the average distance between mesophyll cell surfaces within a leaf and the inner surface of stomatal pores. Consistent application of the conductance model would thus require measurements on leaf cross-sectional images. Parkhurst (6) postulated that intercellular air space is a very important, usually ignored, component of the leaf gas exchange pathway, although it is more relevant for photosynthesis than for transpiration.

Calculations with the electric analogy are simplified by replacing all conductances with resistances, taking $r_w^i = (g_w^i)^{-1}$, where the superscript “i” stands for “bl,” “st,” “cut,” or “ias” (boundary layer, stomata, cuticle, and intercellular air space, respectively). Figure 3 presents the corresponding equivalent electrical circuit. The cuticular resistance acts as a resistance in parallel with the series

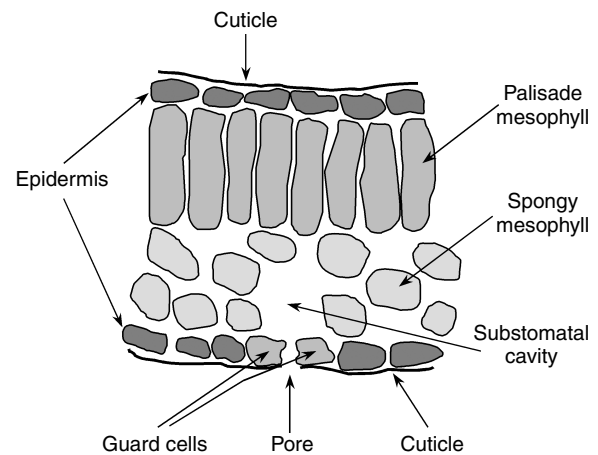


Figure 2. Diagram of a (hypostomatous) leaf cross section, showing different tissues and cell types.

sum of the stomatal and intercellular space resistances. According to this scheme, the total leaf resistance to water movement can be described as

$$r_w^{\ell} = \frac{(r_w^{\text{ias}} + r_w^{\text{st}}) \cdot r_w^{\text{cut}}}{r_w^{\text{ias}} + r_w^{\text{st}} + r_w^{\text{cut}}} + r_w^{\text{bl}}$$

and the total leaf conductance is

$$g_w^{\ell} = (r_w^{\ell})^{-1}$$

Nobel (2) provided some representative conductance values: for crop plants such as beet, spinach, tomato, and pea, g_w^{ias} ranges from 800 to 4000 mmol/m²/s (20 to 100 mm/s); for open stomata, g_w^{st} ranges from 200 to 800 mmol/m²/s (5 to 20 mm/s). These values are somewhat lower for other agricultural crops.

THE STUDY OF STOMATA

Malpighi saw pores on leaf surfaces in 1674. Grew showed them in his anatomical work of 1682. The term “stomata”

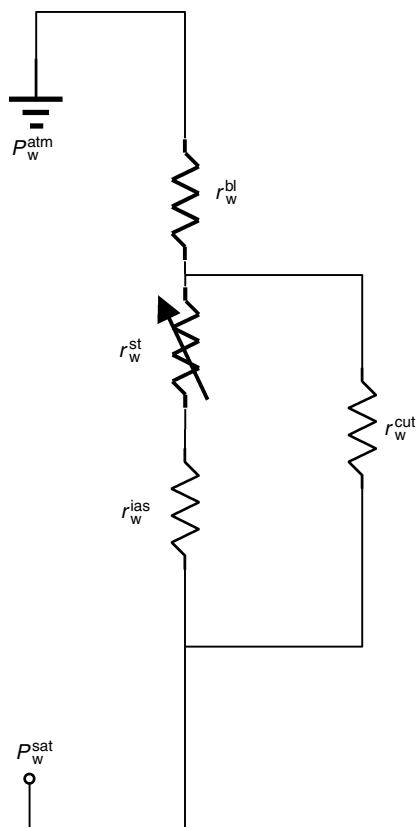


Figure 3. Conductance model of leaf water vapor movement, expressed in resistance form using electrical symbols. P_w^{atm} and P_w^{sat} are the water vapor pressure in the atmosphere and on the surface of mesophyll cells, respectively. The r_w^i are the resistances to water vapor movement in the boundary layer ($i = \text{"bl"}$), in stomata ("st"), through the cuticle ("cut"), and in the intercellular air spaces ("ias").

may have been first used by A. de Candolle in 1827, and research on stomatal behavior began earnestly in the mid-1900s (1). Gas exchange through small pores such as stomata was explained formally in physical terms by Brown and Escombe (7). Later, Stålfelt (8) and Bange (9) showed how, in moving air (when boundary resistance is low), transpiration and stomatal aperture are closely correlated. More recently, Zeiger (10) reviewed the multiple environmental influences on stomatal behavior; the review by Outlaw (11) shows the current state of the art.

The crucial role of stomata in controlling plant CO_2 uptake and water loss has motivated the search for methods of measuring stomatal aperture and conductance. Weyers and Meidner (12) extensively reviewed these techniques. A few salient methods are discussed briefly below.

Microscopy

In some plants, pieces of epidermis can be peeled off the leaf and observed on the microscope after fixing in alcohol. However, it is very difficult to peel the epidermis from some plants, and the process can also affect stomatal aperture. An alternative is to make impressions of the epidermis while the leaves are still on the plant, using

substances such as collodion, silicone rubber, dental paste, or nail polish. These impressions are later peeled off and examined on the microscope (1).

The above mentioned methods involving epidermal manipulation may not be fully representative of *in vivo* conditions because the material used to make the impression may invade the pore or stomatal aperture may change during the process. There is evidence that ultrafast freezing can prevent aperture changes (13). This, coupled with scanning electron microscopy, can capture stomatal characteristics in great detail (e.g., 14).

Anatomically Explicit Models

Most implementations of conductance models assume that diffusion paths for both CO_2 and water vapor are identical, thus making transpiration and photosynthesis proportional to each other as stomatal aperture changes. This assumption ignores the intracellular, liquid-phase components of the CO_2 pathway, although Nobel (2) postulated that liquid-phase conductance values are significant.

Parkhurst (6) criticized conductance models "in part because they are usually one-dimensional representations, but also because they treat continuously interacting processes as if they were sequential." The lumped-parameter conductance approach, as opposed to using spatially distributed parameters and processes, may introduce error even in the apparently simple simulation of diffusion within intercellular spaces, especially when estimating photosynthesis: assimilation really occurs throughout the tortuous intercellular space concurrently with a gradual fall in CO_2 concentration, rather than lumped at the end of the flow path as in typical conductance models.

However, lumping the properties of the leaf interior was unavoidable until recent developments in numerical methods made it possible to solve diffusion equations in very complex domains. Parkhurst (6) proposed a complex distributed electrical analog model, and Pachepsky and Acock (15) created a very sophisticated finite-element model, 2DLEAF, that explicitly represents leaf tissues in a two-dimensional domain based on measurements made on micrographs of epidermal peels and leaf cross sections. Driven by environmental variables such as photon flux density, air and leaf temperature, gas concentrations in the atmosphere, and a novel form of parameterizing the leaf boundary layer (16), the model then simulates the fluxes of water vapor, CO_2 , and O_2 . Calculations with 2DLEAF have shown that distributions of water vapor and CO_2 concentrations in the leaf interior differ from each other, that photosynthesis and transpiration rates are not proportional to each other, and that stomatal aperture affects transpiration more than it affects photosynthesis (15); this confirms the statement by Parkhurst (6) that low stomatal conductance reduces water loss more than it reduces CO_2 uptake.

Infiltration

Another experimental technique for estimating stomatal aperture involves infiltrating graded solutions of different

viscosity into leaves. The viscosity of the most viscous solution that can infiltrate the pores serves as a measure of aperture (17). However, this method does not allow calculating stomatal resistances, and, due to the differences in stomatal anatomy among species, it is better suited to relative measurements of stomatal aperture within the same species (3).

Mass or Viscous-Flow Porometers

The combined resistance of the stomata and the intercellular air spaces of the mesophyll can be estimated for an amphistomatous leaf by forcing pressurized air through the leaf from one surface to the other and then measuring the airflow rate (or the time needed for a specified drop in pressure). The flow rate is inversely proportional to the viscous-flow resistance.

Many such mass- or viscous-flow measuring devices, or porometers, have been described in the literature (18–20). However, absolute calibration is difficult because the flow path inside the leaf is complex and the series circuit measured by the instrument (air flows in through one leaf surface, the mesophyll, and out the other surface) is not representative of the actual water vapor diffusion paths, where the stomata of the two leaf surfaces are in parallel (3). Additionally, mass flow porometer designs have been cumbersome (albeit ingenious), and contemporary interest in separating the adaxial, abaxial, and mesophyll resistance components has decreased interest in mass flow instruments in favor of the diffusion porometer.

Diffusion Porometers

More advanced porometers generally include small cuvettes or chambers that can be attached to leaves and either measure the time necessary for a specified change in the humidity in the chamber (transit-time diffusion porometers; 21) or measure the water vapor concentration differential between the air leaving the chamber and the air entering it (continuous-flow diffusion porometers; 22). They can work with one or both leaf surfaces.

Diffusion porometers are powerful, albeit expensive. They are portable, operate rapidly (seconds to a few minutes), and store data in an onboard memory for subsequent downloading to a personal computer. Although they can work with individual leaf surfaces (unlike mass-flow porometers), they must still average over a relatively large leaf area and thus cannot describe the behavior of individual stomata. They are also affected by any factor that influences water loss (e.g., cuticle conductance), not just stomatal conductance.

Jones (3) suggested the following equation to estimate leaf conductance from a continuous-flow porometer:

$$g_w = (r_{\ell w} + r_{aw})^{-1} = \frac{u_e(x_{w0} - x_{we})[1 - (x_{ws} + x_{w0})/2]}{A(1 - x_{w0})(x_{ws} - x_{w0})}$$

where g_w is the total conductance to water vapor ($\text{mol/m}^2/\text{s}$); $r_{\ell w}$ is the leaf (combined stomata, cuticle, intercellular air space, and cell wall) resistance to water vapor ($\text{m}^2 \text{ s/mol}$); r_{aw} is the boundary layer resistance to water vapor, typically specified by the porometer

manufacturer; u_e is the flow rate entering the chamber (mol/s); x_{we} is the mole fraction of water vapor in the air entering the chamber (mol/mol); x_{w0} is the mole fraction of water vapor in the air leaving the chamber (mol/mol); x_{ws} is the saturation mole fraction of water vapor at the chamber temperature (mol/mol), where the relative humidity of the outlet air is equal to x_{w0}/x_{ws} ; and A is the surface area of the leaf exposed in the chamber (m^2).

STOMATAL FUNCTION

Stomata open and close due to changes in the shape of guard cells. Changes in the concentration of ions inside guard cells make water move into or out of the cells osmotically. The corresponding changes in hydrostatic pressure (turgor pressure, the pressure exerted by the fluid in the cell that presses the plasma membrane against the cell wall) in turn make the cells swell or shrink. Two additional factors influence the change in shape; first, guard cell walls have a set of cellulose microfibrils arranged radially around the cell (radial micellation). When the cell volume tends to increase with turgor pressure, the microfibrils force a lengthening of the cells rather than an increase in diameter. Additionally, the inner walls of guard cells, those adjacent to the pore, are more rigid than the outer walls. Thus, a lengthening of the cell translates into an increase in its curvature, and consequently into an opening of the stoma (Fig. 1b).

The intake of ions for stomatal opening occurs through the guard cell membrane. A special form of enzyme called proton ATPase removes protons from the inside of the cell. This creates an electrical gradient across the membrane; potassium ions can move into the cell (through specialized ion channel proteins embedded in the membrane) driven by this gradient. Chlorine and calcium ions, as well as starch, play additional roles in this process. For further reading, Nicholls and Ferguson (23) presented an excellent introduction to ATPases and active transport through membranes; Mansfield et al. (24) discussed the role of calcium.

Stomatal aperture responds to numerous environmental factors such as light, temperature, relative humidity, CO_2 concentration, and pollutants. It also depends on internal variables such as water stress, periodic metabolic oscillations (circadian rhythms), and the concentration of signaling substances such as abscisic acid (ABA) and cytokinins. Outlaw (11) and Zeiger (10) explained all of this in detail; a brief outline is given below.

In general, stomata tend to open in the light, when there is energy available for photosynthesis. An interesting exception is that of desert succulent plants of the crassulacean acid metabolism (CAM) type. They open their stomata at night (when atmospheric water demand is low), store CO_2 as malic acid, and then convert that into sugars via photosynthesis during the day with their stomata closed and the plant safe from water loss.

Stomatal aperture is generally inversely proportional to CO_2 concentration. Some researchers believe that plants sense the concentration of CO_2 in the leaf interior and respond to that; others believe that the plant responds to the ratio of internal to external CO_2 concentrations.

Stomata tend to close at midday, suggesting that plants can either sense atmospheric humidity, or sense transpiration rate and react to limit water loss. Temperature also affects stomatal aperture differently for different species; in general, temperature extremes close the stomata. However, it is difficult to separate the high temperature effect from that of a high vapor pressure deficit (i.e., high water demand). Wind effects are complex. On one hand, wind reduces boundary layer resistance and thus tends to increase transpiration. On the other, there are reports of plants closing their stomata in response to wind or vibration.

ABA is a plant hormone believed to be the main signaling mechanism for stomatal control. Roots subjected to dry soil (or hypoxia due to flooded soil) produce ABA, which moves upward to the leaves through plant vascular tissues. ABA interacts with guard cells, allowing the membranes to leak potassium ions out of the cell, reducing its concentration gradient, which is accompanied by a loss of water and turgor pressure and subsequently, stomatal closure. Tal and Imber (25) presented an interesting experimental confirmation of the central role of ABA in stomatal control. They showed that mutants that cannot synthesize ABA show permanent wilting because they cannot close their stomata to stop water loss. When ABA is applied to these plants from an external source, however, the stomata close, turgor pressure is restored, and wilting stops.

STOMATAL REGULATION

Stomatal aperture provides plants with a control mechanism with which they can balance the entry of the CO₂ used for photosynthesis and the water lost by diffusion through the pore from evaporation from the surfaces of cells inside the leaf (2). Transpiration is usually considered an inevitable, undesirable side effect of CO₂ uptake. However, transpiration does have important roles: transpiration provides energy to transport water (and hence, nutrients and signaling substances) in the plant, and it also dissipates the heat of direct sunlight (by evaporative cooling).

Many ideas have been proposed to explain stomatal behavior in terms of optimization of cost (transpiration) and benefits (photosynthesis). Optimally, stomatal aperture would vary throughout the day such that there is maximum photosynthesis and minimum transpiration. This implies a complex control system that monitors several input variables (several of which were mentioned in the previous section). Jones (3) and Nobel (2) provided interesting conceptual discussions in terms of feedback and feedforward systems. The latter explains the popular hypothesis that stomata behave so that the sensitivity of photosynthesis and transpiration to stomatal aperture (i.e., the corresponding partial derivatives with respect to stomatal aperture) are a constant multiple of each other. Cowan (26) and Farquhar and Sharkey (27) discussed this and other optimization strategies in greater detail.

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CROP WATER STRESS DETECTION USING REMOTE SENSING

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Remote sensing in the optical spectrum has potential to detect plant manifestations of both transient and chronic crop water stress. Crop water status and evaporation rate have been directly related to thermal indexes, such as the crop water stress index (CWSI), the water deficit index (WDI), and the thermal kinetic window (TKW) index. Measurements of surface reflectance in narrow visible and near-infrared wavelengths have potential for monitoring crop photosynthesis and fluorescence. Surface reflectances in wide bands of the visible, near-IR, and short-wave IR spectrum have been used to detect plant manifestations of chronic crop water stress, such as phenological adaptations and leaf expansion and loss. To apply these approaches with satellite-based sensors for farm-scale irrigation scheduling and crop management, there are still needs for improved sensor design, timely image delivery, and reduced image cost.

INTRODUCTION

Numerous studies have shown that crop water stress has a direct effect on crop growth, development, yield, and ultimately, on farmers' profits. New management techniques in arid and semiarid regions are designed for controlled deficit irrigation to ensure low water loss without yield reduction. To achieve this delicate balance between water use and crop yield, farm managers need an operational means to quantify plant water deficit and evaluate the effects of stress on a given crop species at any stage of development. The following sections present the plant manifestations of water stress that can be detected using passive remote sensing methods with sensors that

acquire data in the visible (VIS), near-infrared (NIR), short-wave infrared (SWIR), and thermal infrared (TIR).

CROP ADAPTATIONS TO WATER STRESS

Turner (1) provided a good synthesis of the effects of water deficit on crop plants and physiological adaptations to transient stress. He identified crop physiological adaptations associated with three types of drought resistance—drought escape, drought tolerance at high water potential, and drought tolerance at low water potential—where “drought” was defined as a period without significant rainfall. The following discussion focuses on the physiological adaptations that might have the greatest effect on the reflective and thermal spectral regions and on the optical properties of plants that would allow stress detection by remote sensing.

Drought escape is the ability of the plant to complete its life cycle before serious soil and plant water deficits develop. For example, studies have shown that wheat can hasten maturity in response to mild water deficits at the critical time between flowering and maturity. When stress occurred between floral initiation and wheat ear emergence, the number of wheat tillers that produced ears was less than that under well-watered conditions.

Drought tolerance at high tissue water potential is sometimes referred to as drought avoidance; it allows plants to endure drought periods while maintaining high plant water status. One such crop adaptation is the reduction of water loss through increased stomatal and cuticular resistance. This is expressed in increased epidermal waxes of leaves and a reduction in general plant productivity. Another adaptation is to reduce the radiation absorbed by the plant through leaf movement (e.g., leaf cupping, paraheliotropism, or wilting) or increased leaf pubescence and waxiness. Drought tolerance is also achieved by reduction of leaf area through decreased leaf expansion, reduced tillering and branching, and leaf shedding.

Water stress is often expressed by the inability of some crop species to maintain cell turgor at low water potentials. The capability of crops to control cell turgor varies widely by species (1). These variations in cell turgor under drought conditions can influence the closure of stomatal apertures and the rates of photosynthesis, evaporation, and leaf expansion.

Optical Properties of Plant Leaves

A great deal has been written on the optical properties of crop leaves in the VIS, NIR, SWIR, and TIR domains (2). In the visible domain, leaf reflectance is affected mainly by leaf pigments such as chlorophyll, xanthophyll, carotenoids, and anthocyanins, as well as by the leaf structure and the dry matter content. In the NIR, reflectance depends on the anatomic structure of the leaves and increases with the number of cell layers and the size of the cells. In the SWIR, reflectance is mainly affected by the leaf water content; strong water absorption bands are at 1.45, 1.95, and 2.5 μm . It is generally reported that leaves under water stress show a decrease in reflectance

in the NIR spectrum, a reduced red absorption in the chlorophyll active band (0.68 μm), and a consequent blue shift of the red edge (3). In the TIR, there is a direct link between the process of plant water evaporation and the plant thermal response (i.e., water evaporates and cools the leaves).

The theory to explain these leaf optical properties and the effect of leaf constituents on leaf reflectance and transmittance was developed into a leaf model by Allen and Richardson (4). The PROSPECT model (5), based on a plate model for multiple layers, is widely used to simulate leaf optical properties as function of leaf chlorophyll concentration (C_{ab}), equivalent water thickness, dry matter content, and leaf internal structure; it enables simulation of leaf reflectance and transmittance as a function of different stresses.

Optical Properties of Plant Canopies

Though knowledge of the optical properties of individual leaves contributes to our understanding of the processes involved, field studies have shown that leaf optical properties alone are not sufficient to understand canopy-level reflectance. Canopy reflectance is a function of the leaf optical properties previously described, plus viewing and illumination geometry, canopy structure, and soil reflectance. For example, Guyot et al. (6) reported that the most important factor that influences the reflectance of a plant canopy is its geometric structure, not its leaf reflectance. Jackson and Ezra (7) concluded that water stress-induced changes in visible, NIR, and SWIR reflectance of a cotton canopy were due largely to canopy geometry changes rather than leaf physiological/anatomic changes in all but the red spectral band.

Furthermore, Myers et al. (8) identified seven parameters (in addition to leaf reflectance) that determined crop canopy reflectance, of which only the first three could be directly related to crop stress: (1) transmittance of leaves, (2) amount and arrangement of leaves, (3) characteristics of other components of the vegetation canopy, (4) characteristics of the background, (5) solar zenith angle, (6) look angle, and (7) solar azimuth angle. Canopy reflectance models enable the simulation of these canopy structural characteristics and viewing geometry on canopy-level reflectance. Closed agricultural canopies can easily be assumed to be horizontally homogeneous, plane-parallel turbid mediums, and therefore the plate model theory can be applied successfully. On the other hand, open crop canopies, showing heterogeneous structure with different shapes, need the development of models that specifically treat the effects of openings, shadows, and canopy geometry on the modeled reflectance. A detailed discussion of the different canopy reflectance model approaches by turbid-medium, geometric, hybrid, and ray-tracing methods is provided by Goel (9).

Similarly, the TIR emittance of a plant canopy is a function of the temperatures of both the plant components and the soil. T_c is the canopy temperature, defined by Norman et al. (10) as the TIR temperature in which the "vegetation dominates the [measurement] field of view minimizing the effect of soil." T_o is the temperature of the soil surface. T_s is the surface

composite temperature, defined by Norman et al. (10) as the "aggregate temperature of all objects comprising the surface," which was shown by Kustas et al. (11) to be a linear function of T_c and T_o .

REMOTE SENSING FOR DETECTING CROP WATER STRESS

During the past 30 years, progress has been made in using remotely sensed data to retrieve information useful for irrigation scheduling and management. The basic approaches have focused on parameters directly related to crop water status (e.g., crop water loss, evaporation, metabolism, photosynthesis, and fluorescence) and plant manifestations of chronic crop water stress (e.g., phenology and leaf expansion and loss).

Crop Evaporation

An important breakthrough in using remote sensing for water stress detection was the development of the Idso-Jackson crop water stress index (CWSI) (12,13). Jackson et al. (12) derived the theoretical CWSI ($CWSI_t$) by relating the canopy-air temperature difference ($T_c - T_a$) to net radiation (R_n), vapor pressure deficit (VPD), soil heat flux (G), volumetric heat capacity of air (ρC_p), psychrometric parameters (γ and Δ), and canopy and aerodynamic resistances (r_c and r_a), where

$$(T_c - T_a) = [r_a(R_n - G)/\rho C_p] \{ \gamma(1 + r_c/r_a) / [\Delta + \gamma(1 + r_c/r_a)] \} - \{ VPD / [\Delta + \gamma(1 + r_c/r_a)] \} \quad (1)$$

Taking the ratio of actual (E for any r_c) to potential (E_p for $r_c = r_{cp}$) crop evaporation rates gives

$$E/E_p = [\Delta + \gamma^*] / [\Delta + \gamma(1 + r_c/r_a)] \quad (2)$$

where r_{cp} is the canopy resistance at potential evaporation. Jackson et al. (12) defined the $CWSI_t$, ranging from 0 (ample water) to 1 (maximum stress), as

$$CWSI_t = 1 - E/E_p = [\gamma(1 + r_c/r_a) - \gamma^*] / [\Delta + \gamma(1 + r_c/r_a)] \quad (3)$$

Though Jackson et al. (12) provided a thorough theoretical approach for computing CWSI, the concept is more universally applied using a semiempirical variation proposed by Idso et al. (13) based on the "non-water-stressed baseline." This baseline is defined by the relation between ($T_c - T_a$) and VPD under nonlimiting soil moisture conditions, example, when the plant water is evaporating at the potential rate (Fig. 1). Such non-water-stressed baselines have been determined for many different crops, including aquatic plants and both preheading and postheading growth periods for small grains (14).

Application of CWSI with satellite- or aircraft-based measurements of surface temperature is generally restricted to almost full-canopy conditions, so that the surface temperature sensed is equal to the canopy temperature (15). To deal with partial plant cover

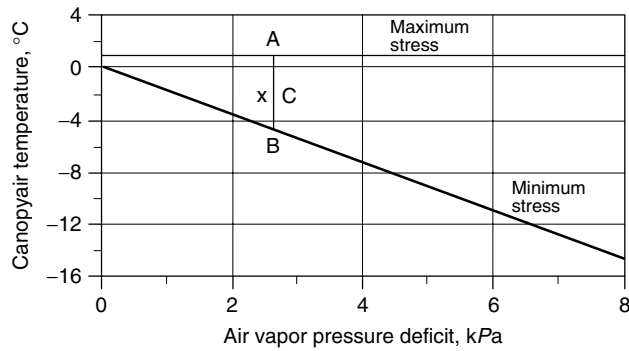


Figure 1. Canopy minus air temperature ($T_c - T_a$) versus vapor pressure deficit (VPD) for well-watered (line labeled minimum stress) and maximally stressed alfalfa based on measurements at various sites across the United States. The CWSI for a measured alfalfa canopy is computed as the ratio of the distances CB and AB (13).

conditions, Moran et al. (16) developed a Water Deficit Index (WDI) which combined measurements of reflectance with surface temperature measurements (a composite of both the soil and plant temperatures), as expressed by

$$\text{WDI} = 1 - E/E_p = [(T_s - T_a)_m - (T_s - T_a)_r] / [(T_s - T_a)_m - (T_s - T_a)_x] \quad (4)$$

where subscripts m, x, and r refer to minimum, maximum, and actual, respectively. The WDI is operationally equivalent to the CWSI for full-cover canopies, where $T_s = T_c$. Graphically, WDI is equal to the ratio of distances AC/AB in the trapezoidal shape presented in Fig. 2, where $\text{WDI} = 0.0$ for well-watered conditions and $\text{WDI} = 1.0$ for maximum stress conditions. The left edge of the Vegetation Index/Temperature (VIT) trapezoid corresponds to $T_s - T_a$ values for surfaces evaporating at the potential rate; the right edge corresponds to $T_s - T_a$ values for surfaces in which no evaporation is occurring. In practice, WDI uses the Penman–Monteith energy balance equation to define the four vertices of the VIT Trapezoid that encompasses all possible combinations of a spectral vegetation index (e.g., Normalized Difference Vegetation Index or NDVI) and $T_s - T_a$ for one crop type on one day (Fig. 2).

Another promising approach for operational application is the use of remotely sensed crop coefficients (the ratio of actual crop evaporation to that of a reference crop) for estimating an actual, site-specific crop evaporation rate from readily available meteorologic information (17,18). This approach requires only a measure of NDVI and is simply an improvement of an approach already accepted and used by farmers to manage crops, where such improvements include increases in the accuracy of evaporation estimates and, by using images, the ability to map within-field and between-field variations.

Crop Metabolism

Quite distinct from the CWSI and WDI, Burke et al. (19) developed a concept of thermal stress in plants that linked the biochemical characteristics of a plant with its optimal leaf temperature range. The thermal kinetic

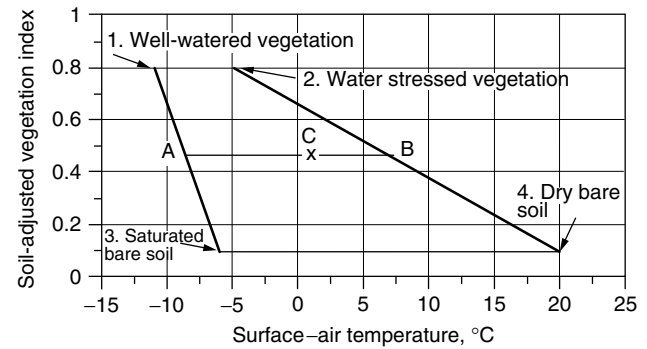


Figure 2. The trapezoidal shape that would result from the relation between surface temperature minus air temperature ($T_s - T_a$) and the soil-adjusted vegetation index (SAVI). Using a measurement of $T_s - T_a$ at point C, the ratio of actual to potential evaporation (which is equivalent to the Water Deficit Index or WDI) is equal to the ratio of the distances CB and AB (16).

window (TKW) is the range of temperatures within which the plant maintains optimal metabolism. For example, the TKW for cotton growth is 23.5 to 32°C; the optimum temperature is 28°C, and biomass production is directly related to the amount of time that canopy temperatures are within the TKW (Fig. 3), provided that insolation, soil moisture, and nutrients are nonlimiting. TKWs have been identified for several crop species, including cotton, wheat, soybean, potato, cucumber, and bell pepper. In practice, the TKW provides a biological indicator of plant health that is being used for irrigation management.

Crop Photosynthesis

Measurements of leaf chlorosis, which causes diminished leaf photosynthetic capacity, may be particularly suitable for early water stress detection. Differences in remote sensing reflectance due to changes in chlorophyll concentration levels have been detected in the green peak and along the red edge spectral region (690 to 750 nm) (20). The red edge occurs between the wavelengths of 0.69 and 0.76 μm due to the change in reflectance caused by chlorophyll absorption in the red spectrum and multiple scattering from leaves in the NIR spectrum (21) (Fig. 4). Several hyperspectral

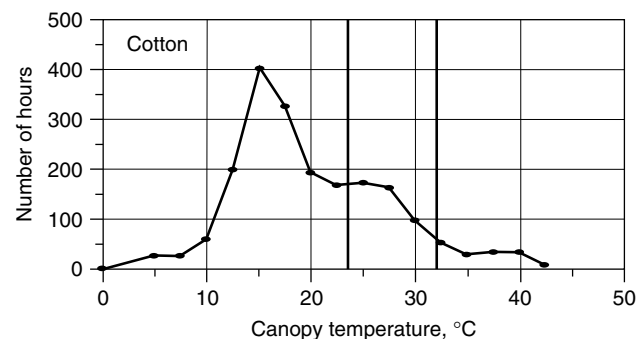


Figure 3. A frequency distribution of seasonal canopy temperatures of cotton, where the vertical lines represent the temperature range that comprises the species-specific thermal kinetic window (TKW) as determined from the changes in the apparent K_m with temperature (19).

indexes proposed in the literature track and quantify chlorophyll concentration for both leaf and canopy-level studies (22,23), allowing remote sensing detection methods to identify vegetation stress through the influence of chlorophyll content variation. These narrow-band vegetation indexes are, among others, the Simple Ratio Pigment Index, SRPI (ρ_{430}/ρ_{680}); Normalized Pheophytinization Index, NPQI ($(\rho_{415} - \rho_{435})/(\rho_{415} + \rho_{435})$); Photochemical Reflectance Index, PRI ($(\rho_{531} - \rho_{570})/(\rho_{531} + \rho_{570})$); Normalized Pigment Chlorophyll Index, NPCI ($(\rho_{680} - \rho_{430})/(\rho_{680} + \rho_{430})$); Greenness Index, G (ρ_{554}/ρ_{677}); Structure Intensive Pigment Index, SIPI ($(\rho_{800} - \rho_{450})/(\rho_{800} + \rho_{650})$); and red edge reflectance-ratio indexes such as (ρ_{740}/ρ_{720}) , $(\rho_{734} - \rho_{747})/(\rho_{715} + \rho_{726})$, (ρ_{750}/ρ_{700}) , and (ρ_{750}/ρ_{710}) . See a review by Zarco-Tejada et al. (23) for more detail.

In agricultural canopies that have large spectral contributions from the soil background and LAI variation in different growth stages, combined indexes are proposed to minimize background soil effects while maximizing the sensitivity to chlorophyll concentration. CARI (chlorophyll absorption in reflectance index) (24) and MCARI (modified chlorophyll absorption in reflectance index) (25) have been proposed for combination with SAVI (soil-adjusted vegetation index) (26) and OSAVI (optimized soil-adjusted vegetation index) (27) to reduce background reflectance contributions. Successful chlorophyll estimation for corn canopies at different growth stages was achieved using the TCARI/OSAVI combined index (28) and with MCARI/OSAVI on open-tree crop canopies (29).

New developments related to vegetation stress, that have potential application to water stress detection, focus on monitoring chlorophyll fluorescence (CF). Red and far-red light is emitted from photosynthetic green plant tissues in response to photosynthetically active radiation. Several reviews offer details on theory, measurement methods, and interpretation, as well as the relationship to photosynthesis, plant physiological status, and photosynthetic functioning (30–32). In particular, relationships between steady-state and dark-adapted chlorophyll fluorescence features with water stress status have been investigated for both active and passive sensors (33–37). This suggests the potential for stress detection using far-distance remote sensing instruments.

Steady-state CF and photosynthetic rates are inversely related, such that CF is low when photosynthesis is high,

although it can be reversed because of an intensified protective quenching action on CF production. The total amount of CF emitted by Photosystems I (PS-I) and II (PS-II) is typically less than 5% of total light absorbed. Thus, accurate remote detection of natural CF would enable the measurement of canopy-level CF without direct contact with the vegetation canopy. Zarco-Tejada et al. (38) demonstrated that CF could be detected at both leaf and canopy levels using multiple approaches, including observing its effects on the red edge spectral region.

Crop Phenology

There is evidence that chronic crop water stress can either hasten (1) or delay (39) crop development, depending on the species involved and the timing of water stress. The timing and duration of stress is also of critical importance to ultimate yield; example, if intense water stress occurs during anthesis of a wheat crop, the impact on final grain yield is usually much greater than if this same stress occurs earlier or later in crop development. Tucker et al. (40) showed that crop phenological stage could be determined using a combination of spectral data and accumulated temperature units (growing degree day). Idso et al. (39) equated the slope of the spectral vegetation index (VI) over time with the rate of senescence and correlated this slope with final grain yield for wheat and barley under stressed and nonstressed conditions (Fig. 5). In a similar study, Fernández et al. (41) found that the hydric stress on wheat could be determined by the slope of the NDVI with time.

Chronic crop water stress can also manifest itself in reduced leaf expansion and leaf loss. There have been several studies that have monitored stress-induced reductions in biomass and GLAI using remote sensing techniques (42–44). Though these results are encouraging, this approach is limited by the fact that the VI–GLAI relation is exponential, leading to saturation of the NIR response at GLAI values of five to six (2). A better measure of the effects of water stress on leaf expansion and biomass production might be the amount of solar radiation intercepted, which is related directly to plant growth. Kumar and Monteith (45) showed that the fraction of absorbed photosynthetically active radiation (fAPAR) is related linearly to NIR/red reflectance. More recent work

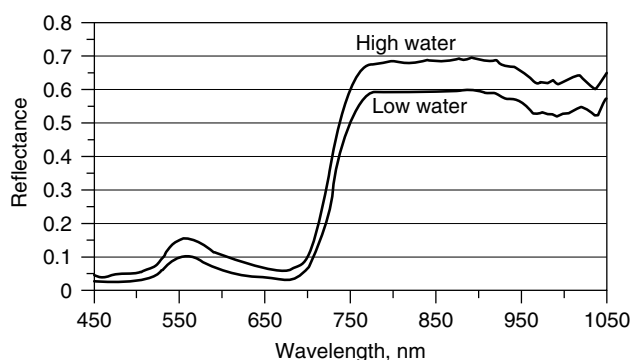


Figure 4. The expected trends of spectral reflectance of a crop canopy with high and low water treatments (21).

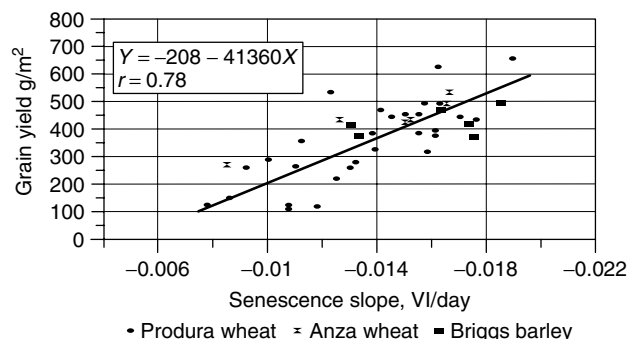


Figure 5. Final grain yield of Produra wheat, Anza wheat, and Briggs barley versus the slope of the transformed vegetation index (TVI6) during the senescence period (39).

by Pinter (46) has shown that this relation between VI and fAPAR is independent of variations in the solar zenith angle, thus increasing the usefulness of this remote sensing approach.

CONCLUDING REMARKS

The CWSI, WDI, and TKW are good examples of remote sensing indexes that link surface temperature measurements to crop and soil evaporation rates. Quantification of leaf chlorosis based on surface reflectance in narrow wavelength bands also has potential for detecting transient water stress. Season-long measurements of surface reflectance in wide wavelength bands have been used to detect the effects of chronic plant stress, such as delayed (or hastened) crop phenology and leaf expansion or loss. These are only some examples of the many approaches available for detecting crop water stress using remote sensors. However, to realize the full potential of satellite-based estimation of crop water stress, it will be necessary to continue sensor development, improve image availability and timely delivery, and reduce image cost.

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VACUUM GAUGE TENSIO METER

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Soil water plays an important role in providing water for plant growth. The vacuum gauge tensiometer is a soil water measuring device that is sensitive to soil water change. The quantity of soil water that adheres around soil particles determines the energy level at which it is being retained or pulled. In other words, the plant root has to exert energy to pull water out of a soil matrix. This energy is also expressed as tension the roots experience in removing the water. Tensiometer is like a mechanical root but shows the tension in the vacuum gauge attached to it. It is an indirect method to determine soil water status. Direct measure of water content is time-consuming and labor-intensive, and the result lags behind the current time. Hence, irrigators depend on indirect methods. Vacuum gauge tensiometer is inexpensive, easy to use, and a preferred measurement method, especially for coarse-textured soils.

A vacuum gauge tensiometer with its components is shown in Fig. 1. Variations from this design exist on the market. A tensiometer is a sealed, water-filled tube equipped with a vacuum gauge on the upper end and a porous ceramic tip on the lower end. The basic components include: a cap, body tube, vacuum gauge with dial, and a porous ceramic tip (Fig. 1).

The Cap: The cap must seal the tensiometer body tube airtight or the device will not work. Some models have a reservoir, like the one shown in Fig. 1, which serves as a water supply source for the body tube. In these models, the cap assembly has an extended stem to hold a neoprene stopper that seals the body tube from outside atmosphere.

Body Tube: The body tube provides support and a liquid connection between a porous tip and the vacuum gauge. Tensiometers come in various lengths. Standard lengths are 6, 12, 18, 24, 36, 48, and 60 inches.

Vacuum Gauge: A hermetically sealed vacuum gauge (Bourdon¹ vacuum gauge) with a round dial is attached

¹A thin-walled flattened tube used in a gauge to read pressure or vacuum developed by Bourdon, an inventor in France some 150

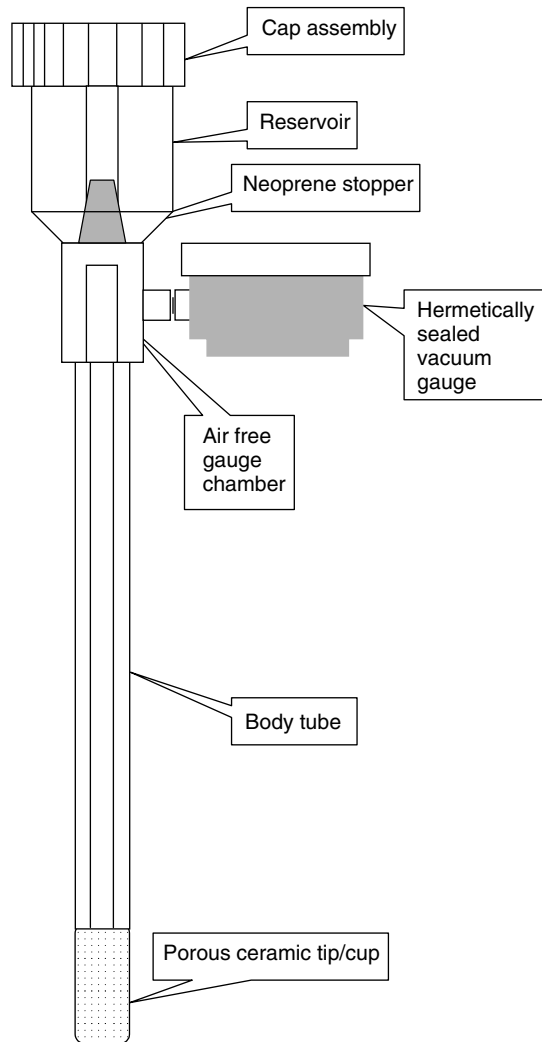


Figure 1. Sketched view of a vacuum gauge tensiometer showing its components.

at the air-free gauge chamber of the body tube. The round dial in the gauge is marked with a scale in centibars to indicate the gauge reading. The scale is calibrated to read in centibar, or hundredths of one "bar." A bar is the unit of pressure, either positive or negative, that has been adopted for the expression of soil water tension. The bar is an international unit of pressure in the metric system and equivalent to 14.5 lb/sq. inch, or 0.987 atmospheres. One centibar is equal to 1 kPa (kilopascal).

Ceramic Tip/Cup: The ceramic tip is built like a tubular cup, and it is porous. The openings of the pores are so small that when saturated with water, air cannot pass through within the range of soil water tensions to be measured. As soil water content outside the porous cup in the soil matrix declines from saturation, water inside the tube starts moving out through the porous tubular tip, which causes a partial vacuum inside the tensiometer that can be read on the vacuum gauge. The needle in the gauge

starts moving above the face of the dial. The dial reading, indicated by the needle, shows the suction or tension at which the water is being pulled by the surrounding soil.

A reading of zero corresponds to a completely saturated condition, regardless of the type of soil. A reading of 80 kPa (centibar) indicates a dry condition for sandy soils and would hurt sensitive crops. This number is also the functional upper limit for tensiometer readings. A tension higher than 85 kPa (centibar) will cause the water inside the tube to vaporize, restricting its use to suctions less than about 85 kPa.

The cup conductance and the gauge sensitivity determine the response time of the tensiometer. Ceramic tensiometer tips/cups for field use generally have a conductance in the order of $3 \times 10^{-5} \text{ cm}^2/\text{s}$, which, in a vacuum gauge system, would have a response time of five seconds. A response time of 1 minute is adequate for most field applications.

The tensiometer is sensitive to very small changes in soil water, making it useful for irrigation scheduling. Irrigation scheduling is a process to determine when to irrigate and how much water to apply. Applying water in an untimely manner with too little or too much water can result in yield reductions. Overirrigation wastes water, costs dollars to pump, and may leach nutrients beyond the root zone. Tensiometers are particularly accurate at low tensions, which is the wettest part of the soil water range. Tensiometers are popular with growers of high-value crops, such as vegetables and fruits, on sandy soils.

A depth label is usually placed on the vacuum gauge or on the side of the tube to indicate the depth at which the ceramic tip will be set when installed, which is important for identification purposes. The soil suction reading on the vacuum gauge dial is an indication of soil water availability for plant use and does not need to be calibrated for salinity or temperature. The readings, however, have different meanings in terms of their use for irrigation scheduling depending on soil type. Table 1 and Fig. 2 provide some interpretation of tensiometer readings in relation to soil texture.

TENSIOMETERS WITH ELECTRONIC READER

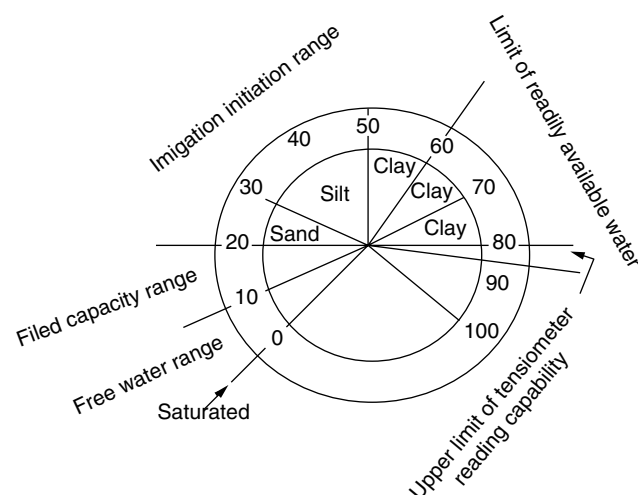
The vacuum gauge tensiometer is not suitable for automated continuous data collection or for controlling irrigation system operation by switching it on or off. However, some models have electronic technology added to the tensiometer that allow them to be remotely read or automatically switch irrigation on or off.

The switching tensiometer models employ the standard vacuum gauge, to which a circular magnet is affixed to the hub of the indicator needle. This magnet is rotated by the action of the indicator needle as it moves according to drying/wetting cycle of the soil. The switch is affixed to the outside of the gauge face and can be adjusted for various readings where the switch will turn "on." The switch contains a reed switch that is held open by a small biasing magnet installed inside the switch. This magnet is of lesser power than the one mounted on the hub of the gauge indicator needle. As the indicator needle of

years ago. It was brought to the United States by Ashcroft Gauge Company 100+ years ago and still is the standard gauge used practically worldwide.

Table 1. Interpretations of Tensiometer Readings

Reading (Centibars)	Status	Explanation/Action
0	Saturated	Soil is saturated regardless of soil type. If readings persist, there is possible danger of waterlogged soils, a high water table, poor drainage and soil aeration, or the continuity of the water column in the tube may have broken.
5–10	Surplus water	Indicate a surplus of water for plant growth. Drainage continues, and persistent readings indicate poor drainage.
10–20	Field Capacity	Field capacity for all types of soils. Additional water will drain as deep percolation carrying nutrients without opportunity for plant use. Sandy soils, however, have very little storage capacity, and suction values increase rapidly as plants remove water past 15 to 20 centibars. For sensitive crops, like potato, rapid irrigation may be required before damaging stress can develop.
20–40	Irrigation range	Available water and aeration good for plant growth in fine- and medium-textured soils. Irrigation is not required for these soils at this range. Coarse-textured soils may require irrigation in the 20 to 30 centibar range and the finer sandy soils at 30 to 40 centibar ranges.
40–60		Usual range for starting irrigation. At 40 to 50 centibar, irrigation may need to be started for loamy soils. On clay soils (silty clay loams, silty clays, etc.), irrigation usually starts from 50 to 60 centibars. Heavy clay soils still have some available water. Irrigation, however, ensures maintaining readily available soil water. The stage of growth and type of crop will influence the decision.
70	Dry	Stress range. However, crop is not necessarily damaged. Some soil water is available in clay soils, but may be low for maximum production.
80		Top range of tensiometer accuracy; higher readings are possible, but tension within the water column inside tensiometer will break between 80 to 85 centibar, which has relationship to elevation of the area compared with mean sea level. At higher elevation, the continuity of water column inside the tube may break at a lower reading because of change in vapor pressure according to change in atmospheric pressure.

**Figure 2.** Schematic of the tensiometer vacuum gauge dial to show interpretation of readings in relation to soil type.

the tensiometer rises with the drying of soil to the point where the switch has been set, the magnet on the gauge overrides the small biasing magnet inside the switch, which causes the reed switch to close. The switch is wired into a low-voltage current loop from the control device to the solenoid valve. The switch closure completes the circuit and allows the solenoid valve to open and irrigation to start. When the soil rewets, the indicator needle drops down and the switch opens interrupting current loop from the control device that causes the solenoid valve to close, terminating irrigation.

For automated data collection, the standard vacuum gauge is replaced by a pressure transducer, where the

partial vacuum reading is converted as current flow signal. The partial vacuum, caused by water moving out of tensiometer, causes a deflection in the face of the transducer, which excites the electronic elements of the device to generate the signal in a certain milliamperage range equivalent to centibar or kPa linear scale. These changing signals coming from the transducer may be recorded by data logger or computer. The reading is continuous. The data acquisition computer may be programmed to start an irrigation system at a certain value. The automated versions of tensiometers enable remote operation of the irrigation system.

WORKING PRINCIPLE OF A TENSIOMETER

Soil water exists primarily as thin films around and between soil particles and is bound to soil particles by strong molecular forces. As the soil dries, the water films become thinner and become more tightly bound to soil matrix. This increase in tension within the films now in contact with the tensiometer causes water to be drawn from the ceramic tip. The withdrawal of water from the ceramic tip creates a partial vacuum within the tensiometer. Water continues to be drawn until the vacuum created inside the tensiometer equals the tension of the water films outside. At this point, equilibrium is reached and water ceases to flow. The vacuum gauge reading indicates the amount of suction or tension.

As water is added to the soil from rainfall or irrigation, the soil suction is reduced. The higher vacuum in the tensiometer causes soil water to be drawn into the tensiometer, and the vacuum will be reduced until a balance in tension is reached. The tensiometer continuously responds and maintains a balance with

the soil water suction or tension, and the vacuum gauge indicates the amount of tension, hence the name tensiometer.

PREPARING THE TENSIOMETER FOR FIELD USE

Preparing a tensiometer for use starts with filling the reservoir and body tube with distilled water, taking care that the ceramic tip is wetted from one direction to avoid air entrapment in the finer pores of the ceramic tip. Allow the tensiometer to stand upright, and soon the tip will wet-up and free water will appear like sweat on the outer surface of the ceramic tip. Refill as necessary to dispel all the air from the tensiometer. A hand-operated vacuum pump may be used to help remove air from the tensiometer. Operating the pump when the tensiometer is filled with water and the tip is submerged in water helps remove gases from the pores of the ceramic tip and solution. After each pumping, refill the tensiometer completely with water and repeat until no more bubbles are observed. The tensiometer is then sealed by screwing the reservoir cap down securely.

After letting it stand in water in a bucket overnight, the tensiometer may be set outside upright in the air. The air will start drying the tip. The gauge should read 70 when the tip is air-dried. Repeat the wetting and drying cycle with the tensiometers that do not respond correctly the first time.

Tensiometers require checking prior to installation. As with any measurement device, proper care and maintenance are required. If the tensiometer was used previously, begin by washing and rinsing it inside and out. Residues on the porous ceramic tip that were not removed by washing may be removed by sanding lightly. Distilled water treated with three to five drops of chlorine bleach per gallon of water can be used to inhibit algae growth. Manufacturers also provide solutions for water treatment, which may be used according to direction. Distilled water available in a grocery store has been found to be adequate. If an excessive amount of air bubbles are noticed, then boiling may be helpful, but the remaining water must be stored in an airtight container.

If tensiometers are not to be installed immediately, cover the tips with a plastic bag to prevent evaporation or let them stand in a bucket of water until installed.

TENSIOMETER INSTALLATION

Site Selection

The number of tensiometer installation sites required will depend on the crops grown and field conditions. Fewer stations of tensiometers are needed when a single crop is grown in large blocks of uniform soil. If the soils are variable, different crops are to be grown and more stations are necessary. Stations need to be selected to represent an area, and care should be taken not to cause excessive compaction or destruction of plants during the installation process.

Except for very shallow-rooted crops, tensiometers are normally installed in groups of two or more to characterize

the soil in the top half to three-quarters of the root zone. If the potential root zone is less than 12 inches, a single tensiometer may be installed in the center of the zone at 6 inches deep. With deeper-rooted crops, one tensiometer should be placed at the upper one-quarter of the rooting depth and another at the lower quarter point or three-quarters of the depth (see Fig. 3).

In deeply rooted crops, or situations where a distinct break in soil textures exists, three or more tensiometers may be needed. An example might be 18 inches of sand overlying a silty soil on which corn is grown and 3 to 4 feet of the root zone is to be managed. One tensiometer might be placed at 6 inches, a second at 18 inches, and a third at 2 to 3 feet. The differences in tension readings would make it possible to better assess the soil water conditions. Almost 70% of crop water is supplied by the top half and only 10% from lower one-fourth of the rooting depth. Irrigators, therefore, often manage only the top half or three-fourths of the root zone. Tensiometers should be long enough to reach the desired depth, and the diaphragm of the vacuum gauge must not touch the ground. The tensiometer should never be set in a hole or a depression.

Tensiometers may be installed using a soil auger or a probe. Manufacturers also provide simple coring tubes. Placement should be in a crop row to avoid traffic. Where furrow irrigation is used, the tensiometers may be angled slightly to place the tip under the furrow. The electronic tensiometers may require a cover to safeguard the electric connections from sprinkler or rainwater. If a valve cover box is used, the tensiometer tips need to be slanted out to be in the crop area. The hole should be small enough to create resistance to insertion of the tensiometer and the bottom shaped to form a close soil contact at the tip, which may be accomplished by returning a little loosened

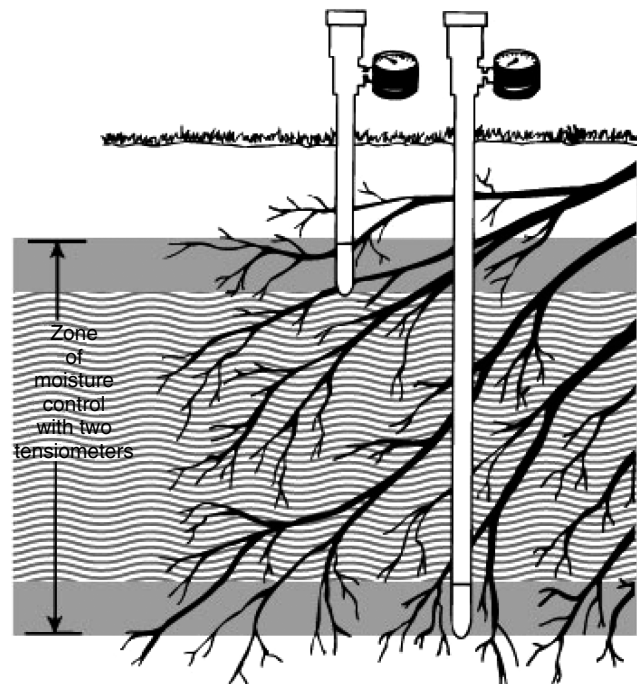


Figure 3. Sketch showing root zone moisture evaluation and control by two tensiometers.

portion of the soil from the depth of placement back into the hole and adding a little water. When the tensiometer is pushed for placement, the soft soil will move around the tip to conform to the rounded shape of the tip and make a good contact. A good contact is absolutely essential for the tensiometer to perform.

Tensiometers must be handled with care, because the tips may break if handled roughly. The depth label on the tensiometer will identify the root zone being monitored. Tensiometer locations need to be marked both in the row and at the edge of the field. A wooden stake painted bright or a metal rod with a colored flag attached are good markers. Locating tensiometers in tall crops can be a problem. A written log of the station locations is also recommended.

Servicing

Tensiometers are weatherproof, except for freezing, and generally require very little service. When first installed, there may be tiny air bubbles clinging to the sides of the body tube. However, after one cycle of soil water use, which creates a high vacuum, the bubbles will rise to the top and can be eliminated by refilling. The amount of bubbles will depend on the gas originally present in the vacuum gauge and the amount dissolved in the water. Servicing is best done soon after irrigation. Tensions are low, and air that may have been drawn into the cup at high tensions can be eliminated by refilling. Tensiometers return to equilibrium rapidly at low tensions. They also respond quickly to a very minute withdrawal of water from the system. If much air is drawn into a tensiometer at low tensions, the porous cup may be defective, and the tensiometer may need to be replaced. Some air entry is unavoidable. When using a number of tensiometers, watch for tensiometers that accumulate abnormal amounts of air. The colored fluid concentrate, supplied by the manufacturer for control of algae, helps to spot air bubbles in the tensiometer more easily.

Tensiometers should be removed from the field before freezing. The water can freeze and break the ceramic tip or the body, or damage the vacuum gauge. Tensiometers need to be emptied before long-term storage, which prevents salt deposition in the porous material with evaporation or rusting of the gauge.

Troubleshooting

A tensiometer that is out of water or leaking will remain at zero on the gauge, or the reading will fluctuate in the low suction range. Two or more successive zero readings may be a sign of a malfunction and should be investigated. If the gauge remains at zero, refill with water and use a hand pump, if available, to remove air. The tensiometer may have been empty because of dry soil. If the tip was dry, fine air bubbles will rise rapidly for several minutes and then cease. If larger bubbles rise and continue, a leak is indicated and the source should be determined. If the bubbles rise from the bottom, remove the tensiometer and replace the tip. If the bubbles enter from the side, the body tube may be cracked and should be fixed. If bubbles rise from the gauge, the leak may be in the gauge or the

threaded connection. A leaky gauge needs to be replaced, but a threaded connection can be resealed. If no large bubbles rise and yet the reading remains at zero then the reservoir cap may be cracked or the seal may be defective. Inspect for 'O' ring if there needs to be one for a proper seal. In most cases, the trouble is easily corrected.

A damaged vacuum gauge may stick in one position or may not respond smoothly with changes in soil water. Check a suspect gauge against one known to be in good working order or replace the suspect gauge with a new one. Readings that are higher than expected, especially after irrigation, generally do not indicate a tensiometer failure. The irrigation water may not have penetrated to the depth of the tensiometer tip.

General Guidelines

- Place two or more tensiometers of different lengths near one another (a station), usually in the crop row. Two stations may be enough in a field with uniform soil and slope. One station should be near the start of an irrigation set and the other at the end of that irrigation set. The location of stations will depend on the type of irrigation system. With furrow irrigation, this location may be at the upper and lower quarter points of the first set in the field. For a center pivot, it may be in the outer and middle spans. If desired, one set may be established at the starting point of the pivot and another set at the endpoint.
- Tensiometer stations should be located in representative areas of the field. Do not position tensiometers in low spots or on knobs, and place them where the plant population is representative of the field.
- Tensiometer installation depth is determined by the active root zone of the crop. For example, for a corn crop on a deep soil, three tensiometers, installed at depths of 12, 24, and 36 inches, are recommended at each station.
- One should wait 24 hours after installing the tensiometer to obtain reliable readings. If the soil was dry at installation, irrigation or rainfall may be needed before obtaining satisfactory readings.
- Tensiometers should be installed as early as possible during the growing season and left in the field for the duration of the growing season. The roots of the crop must grow around the porous tip for reliable readings. Moving of the tensiometer during the growing season is not recommended.

LIMITATIONS OF TENSIO METER

The major criticism of the tensiometer is that it functions reliably only in the wet range of soil water at readings of about 80 centibar or less. At higher readings, the porous tip may leak air and the gases will be drawn out of the water, and at low pressure, the water will vaporize causing discontinuation of the tension column or vacuum. The gauge reading will fall to zero, which is not as serious as it may seem, because most of the available water in coarse-textured soils, and about 50 percent or

more in fine-textured soils, have already been used up at this range.

Another criticism is the price, which ranges from \$35 to \$60. When used in large quantities, the cost may seem prohibitive. Irrigation scheduling, however, has been shown to easily pay off through increased yields or reduced pumping cost. Proper handling may extend the useful life, and the cost may be spread over many seasons, making it cost effective.

Finally, the ceramic tip may gradually fill with precipitates because of soil water movement through the pores, which slows water transfer through the tip and increases the time required for the tensiometer to respond to a change in soil-water conditions. Some slowing does no harm, but if the response time becomes too slow, a new tip should be installed. The response time may be improved by rubbing the exterior of the tip with fine sand paper or soaking the tip in a mild acid solution. The amount of plugging depends on the soil-water chemistry and the manner of use.

Where tensiometers can be left in the ground, the tip porosity remains satisfactory for several years in most soils. However, each time the tensiometer is removed from the soil, tip life is reduced, which is particularly true if the soil is calcareous or saline. In extreme cases, where the tensiometer is installed and removed several times per season, the tip may need to be replaced after one year of use. To minimize this damage, a tip that is removed from the soil should be protected from drying until the tensiometer has been emptied, cleaned, and dried.

TILE DRAINAGE

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Tile (or relief) drains are porous pipes buried approximately 1 m beneath the surface of the soil. These pipes drain excess water from the land into nearby water bodies, most typically cisterns, open ditches, or streams. The most common use of tile drainage is to improve agricultural production by draining excess moisture from the plant root zone and allowing cultivation normally saturated soils. Subsurface tile drainage also lowers the water table and reduces soil salinity. Many of the world's productive agricultural areas, including 25% of the cropland in the United States, require artificial drainage (1). The area drained by subsurface drainage systems in the United States has increased since 1900 from less than 5 to almost 30 million hectares (2).

TILE DRAINS AND SURFACE WATER QUALITY

The impact of tile drainage on receiving water bodies varies (3). Site-specific variables such as soil characteristics, topography, climate, and precipitation patterns as

well as the configuration of the drainage network and other land management practices determine the proportion of precipitation resulting in surface runoff and subsurface drainage and the overall impact on receiving water bodies. Because rainfall can penetrate further into the soil, surface water drainage is reduced in tile-drained fields compared to saturated soils (4). Depending on local conditions, subsurface drainage can reduce surface runoff (4,5).

Increased infiltration prolongs the time that water is in contact with the soil, thereby often reducing the transport of pollutants that are likely to adsorb to soil particles and increasing leaching of pollutants that are easily released (6). The difference between surface and subsurface water drainage may be minimized when there are direct surface water inlets into the subsurface drainage network or during particularly heavy rains, although Schwab et al. (5) found that good subsurface drainage decreased storm peak discharge by as much as 30%. On the other hand, management practices such as riparian buffer strips that are designed to reduce surface runoff usually do not target subsurface drainage. In these cases, tile drains may result in increased pollutant loading during storms (7).

Pesticides and Herbicides

Agricultural pesticides and herbicides enter surface waters by both surface runoff and subsurface drainage. Whether subsurface drainage decreases the amount of pollutant entering the waters depends on the hydrologic characteristics of the site and the particular chemical used. The two pollutant characteristics important to this discussion are persistence (the time it takes to degrade to a nonpollutant form) and soil sorption (the degree to which the pollutant adsorbs to soil particles). Persistent pesticide/herbicides are most likely to drain (either by surface runoff or subsurface drainage) into nearby receiving waters. Subsurface drainage is most likely to reduce the transport of pesticides and herbicides with high soil sorption qualities. Soil characteristics can influence sorption, which is greatest in strongly organic soils and those that have high clay content.

Pesticides are most commonly detected in subsurface flow during the first heavy rain following application. Other peak periods include soil thawing and tillage. However, the most important factors determining pesticide transport are precipitation volume and timing and soil type. Pesticide loss through tile drainage is usually substantially lower than loss through surface runoff. For an excellent review of tile drainage and pesticide transport, see Kladivko et al. (6).

Nutrient Pollution

Nitrogen. Much of the fertilizer applied to a field can be lost through tile drains. This is particularly true for nitrogen, which is more mobile than phosphorus in soil. Different species of nitrogen react differently. Tile drainage usually decreases losses of organic nitrogen and increases loss of nitrates and ammonium (3).

Depending on the form of fertilizer applied, in wet years, up to 70% of the nitrate applied as fertilizer can flow off fields through drainage systems (8). Manure application leads to significantly less nitrate loss to tile drains than commercial fertilizer application (9). Drain placement may also be important for reducing nitrogen losses through drain tiles. For example, shallow drains reduce nitrate leaching (10), and more frequent drain tile spacing reduces ammonium losses (11). Placement of tile drains in areas of preferential flow for manure or fertilizer applications can elevate effluent nutrient concentrations (12).

Phosphorus. Even though phosphorus (P) adsorbs strongly to soil particles, artificial drainage is an important contributor to nonpoint-source phosphorus pollution (13). Burke (14) reported phosphorus losses through drainage of approximately 33% of the amount applied in fertilizer. Other estimates range from 2% (15) to 17% of the applied phosphorus lost in drain tile effluent (16).

The greatest P loss from agricultural systems occurs during intensive rains (17). At these times, both surface flow and tile drain effluent P concentrations are highest (18). Tile drain discharge may be nonlinearly related to rainfall intensity. For example, in a small drainage system in Sweden, Ulen and Persson (19) found that half the yearly P export through drain tiles occurred during an average of 140 hours of episodic rain.

The ability of soils to retain P depends on soil P concentrations. As soils become P saturated, leaching becomes more prevalent. The point at which this happens varies in different soils ranging from 10–119 mg Olsen P kg⁻¹ soil (20). The mass of phosphorus lost from agricultural systems via tile drains is small compared to nitrogen (21), but the impact on phosphorus-limited aquatic systems may be substantial.

TILE DRAINAGE AND MANAGEMENT

Agriculture in many areas would be difficult, if not impossible, without subsurface drainage. Research is ongoing into strategies for managing pollutants in drain tile effluent. Examples include controlled drainage where subsurface drainage systems are fitted with barriers to restrict drainage (3), subirrigation systems that both control drainage and provide increased irrigation to crops during nonsaturated periods (22), constructed treatment wetlands (23), and land retirement (24). Finding solutions that are both environmentally and agriculturally acceptable is critical to the survival of the world's waterways.

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TILE DRAINAGE: IMPACTS, PLANT GROWTH, AND WATER TABLE LEVELS

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Tile drainage is a method of artificial drainage, which lowers the water table and reduces water logging on productive agricultural lands. Improved drainage is an agricultural, rather than a land conversion, practice. The objective is to increase production efficiency, crop yields, and profitability on poorly drained agricultural lands. In the wetter, humid regions, precipitation will cause the water table to rise at, or near, the soil surface. If high amounts of precipitation occur during the growing season, the water table could rise within the crop root zone. The same situation occurs in the drier, arid, and semi-arid regions where irrigation is extensively practiced. Continual irrigation and canal seepage over the years lead to an accretion of the water table, and this could also be injurious to plant growth.

Every soil has some degree of natural drainage. However, if the rates of precipitation or irrigation exceed the natural drainage rate of the soil, the high water table conditions will impair crop growth, because of a lack of oxygen in the root zone. Artificial, or tile drainage, is therefore required to remove excess soil water and to lower the water table, to maintain an optimum soil–air environment within the crop root zone. Most of the world's highly productive agricultural soils require either surface drainage or tile drainage, or a combination of both, to maintain stable crop yields. It is estimated that there are about 200 million hectares of cropland worldwide with improved drainage. The benefits of drainage are given in Fig. 1.

Tile drainage (also known as subsurface pipe drainage) is the installation of horizontal, corrugated, perforated plastic pipes below the soil surface to enable water table drawdown. Parallel lateral pipes are normally of 75 to 100 mm in diameter and are installed on grades varying from 0.1% to 3%. The downstream ends of the laterals

- Ability to better manage and control the water table for enhanced crop productivity
- Reduced flooding
- Better soil water conditions for seedbed preparation and planting
- Earlier and more uniform seed germination
- Improved field machine trafficability
- More favorable air and salt balances in the crop root zone
- Higher crop yields and reduced yield variability
- Ability to plant higher value crops
- Higher farm incomes
- Increased land values
- Less soil erosion
- Better uptake of nutrients
- Improved soil structure
- Enhanced soil microbiological activity
- More timely field machine operations
- Ability to effectively lengthen the growing season, and make better use of lower heat units in northern regions
- Potential to improve water quality

Figure 1. Benefits of tile drainage.

are connected to a buried collector pipe. The diameter of the collector increases with the area drained. The spacing between the lateral pipes depends on the soil type and the amount of water to be removed. The lateral pipes can be installed in a parallel layout, herringbone layout, or at random. Depending on topography, land formation, and proximity of a water receiving body, the collector may outlet by gravity to a watercourse, or into a sump. In the case of the latter, the discharge is then pumped to a lake or stream. Before the advent of plastic pipe in the late 1970s, clay tile pipes were used. However, plastic pipes are lighter and easier and faster to install. The drainage coefficient (or volume of water to be removed by the drainage system) in humid regions is about 10–15 mm/day.

One adaptation of tile drainage is the use of surface inlets to remove water ponded in surface depressions, or as conservation structures to reduce surface runoff and erosion on sloping lands. The drainage coefficient for these systems ranges from 15 to 20 mm/day.

In the irrigated regions, tile drainage not only controls the rise of the water table, but also reclaims salinized soils. The salinity of irrigated soils could increase with time, and this also reduces crop yields. Water in excess of the plant evapotranspiration requirement is applied during irrigation. This additional quantity of water is known as the leaching fraction. Salt is then leached from the root zone and removed from the field via the tile drains. Deeper drain installation ensures that salts do not rise to the root zone because of capillary action, which also prevents waterlogging of the root zone. The amount of surplus irrigation water to be removed by the tile drainage system is less than in humid regions. Consequently, lateral pipe drainage spacings are wider than in the humid regions. The drainage coefficient in irrigated regions varies from 2 to 5 mm/day.

Horizontal pipe drainage systems are usually installed with either a trenchless plow or a chain or wheel type trencher. The trenchless plow is widely used in Europe and North America and results in faster installation rates. The depth and grade of the pipe installation equipment

is achieved with a horizontal laser grade control system. In cases in which installations are conducted in fine sand and silty soils, the pipes are wrapped with a synthetic geotextile, to minimize blockage of the drainage system by the soil particles.

Another form of subsurface drainage is known as vertical drainage or tubewell drainage. It primarily controls waterlogging and salinity in the irrigated regions. The primary purpose of tubewells is to abstract groundwater for irrigation, and consequently, the water table is lowered, and salinization caused by capillarity is minimized. This situation is ideal where the groundwater is not very brackish, or saline, and therefore suitable for irrigation. In areas where the groundwater is saline, then the pumped irrigation water must first be mixed with fresher, or sweeter, water in irrigation canals.

When subsurface pipe drainage reclaims salinized and waterlogged lands, it is viewed as an environmentally beneficial practice, because degraded lands that were once intensively irrigated and cropped are returned to their full productive potential.

Drainage installations result in both hydrologic and land use changes. On poorly drained lands, surplus water is either ponded on fields or moves as surface runoff to nearby open water bodies. On tile drained lands, water percolates below the root zone, which reduces surface ponding and runoff. Infiltrated water is stored in the soil profile and will eventually percolate to the deeper groundwater table.

Some of the most significant technological developments in tile drainage occurred in the period 1965–1975. Beginning in the 1980s, environmental concerns about drainage water quality emerged. High concentrations of nitrates were being measured at the outlet of tile systems. Drainage practitioners responded to these concerns with new technologies such as watertable management, controlled drainage, and subirrigation. Tile drainage is now regarded as a multifunctional water management practice, with systems being designed to not only serve crop needs, but also minimize environmental impacts.

Controlled drainage and subirrigation make use of the existing tile drainage system to raise the water table during certain periods of the year. By either restricting the outflow of water from the tile outlet or by pumping water back into the drainage system, particularly during the growing season, the crop can obtain water from the water table to meet its evapotranspiration requirement. This process helps to reduce the effects of drought stress during the growing season, thus increasing crop yields. An additional important benefit is that the amount of nitrates leaving the tile outlet is reduced, because of a process known as denitrification.

The benefits of tile drainage go beyond the control of excess soil water and salts in the crop root zone. Environmental and socioeconomic benefits are associated with vector control and public health. For example, drainage of stagnant water eliminates malaria, foot-rot in large animals, yellow fever, the liver fluke snail, and other waterborne diseases. The overall benefit is a healthier

human population. Drainage also reduces or eliminates mildew infections and various root rots of plants.

MEASURING AND MODELING TREE AND STAND LEVEL TRANSPIRATION

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SUMMARY OF PRACTICAL IMPLICATIONS

Transpiration is a key process in the application of phytoremediation to soil or groundwater pollutants. To be successful, vegetation must transpire enough water from the soil or groundwater to control or take up the contaminant. Transpiration is driven by a combination of abiotic (climate, soil water availability, and groundwater depth) and biotic (leaf area, stomatal functions, root amount and distribution, and hydraulic characteristics) that need to be evaluated when considering appropriate site and species combinations. The protocols are not trivial, but transpiration can be measured at a variety of scales using techniques such as direct measurements of sap flow on individual trees, eddy flux gradient analyses, or gauged watersheds. Alternatively, models can be used to estimate transpiration, but these usually require on-site calibration or parameterization to produce accurate predictions. Case study analyses across a range of site conditions and species indicate a maximum transpiration capacity of approximately 7.5×10^6 liters of water per hectare per year (8×10^5 gallons of water per acre per year), with a range of 1.5×10^6 to 7.5×10^6 liters per hectare per year (1.6×10^5 to 8×10^5 gallons per acre per year). Variation among sites is related to species, tree size, and stocking (i.e., vegetation density) differences. Application of a physiologically based and site-specific parameterized model suggests reasonable agreement between measured and predicted transpiration estimates for the Air Force Plant 4 site in central Texas.

IMPORTANCE OF ACCURATE MEASUREMENTS OF TRANSPIRATION

Transpiration—the amount of water used by a tree or stand of trees—is one of the key processes in the application of phytoremediation of soil water or groundwater pollutants. To be successful, native or planted vegetation must transpire enough water from the soil or groundwater layer containing the pollutant to control the transport or decrease the mass of contaminant. Hence, quantifying current and future transpiration and determining the principal location of water uptake by native and planted vegetation on the site must be the evaluation criteria for applying phytoremediation. Quantifying transpiration

requires a thorough and accurate assessment of water use patterns such as, transpiration rates, depth of soil water uptake, interactions with climate, and soil water availability. Measuring current transpiration or predicting future transpiration is not trivial. Because transpiration is an integrated response of the atmosphere–plant–soil continuum, measurements and predictions of transpiration capacity must account for (1) variation in climatic driving variables (i.e., solar radiation, water vapor saturation deficit, precipitation, wind speed, and temperature), (2) structural and physiological (leaf stomatal function) characteristics of the vegetation (leaf surface area, and root area and extent), and (3) soil water dynamics (water-holding capacity, and permeability).

Evapotranspiration and transpiration are often used interchangeably, but these processes are different. Evapotranspiration includes the amount of water transpired by the vegetation, and losses due to evaporation of intercepted precipitation and soil surface evaporation. In forests, interception evaporation is a function of rainfall intensity and leaf and branch surface area, ranging from about 10 to 50 percent (1–3). In closed canopied forests, soil evaporation is a minor component of the overall water budget (3), but may become increasingly important in open stands. From a phytoremediation perspective, transpiration is the key factor to consider because interception evaporation does not involve soil water or groundwater.

The process of transpiration involves water movement through the soil, roots, stems, and leaves into the atmosphere in response to water potential gradients—always moving in the direction of smaller potential or negative gradients. Water potential is near zero when water is freely available and decreases to negative values when water becomes more limiting. The movement of water from the leaf interior to the atmosphere occurs through small openings in the leaf called stomata, which open and close in response to external (e.g., climatic factors) and internal (e.g., water potentials of leaves) driving variables. Species vary considerably in stomatal responses to these driving variables and provide opportunities for selecting species to optimize transpiration in different climatic environments.

Five methods are used to quantify transpiration: (1) precipitation minus runoff on gaged watersheds, (2) energy balance (e.g., Penman-Monteith equation), (3) eddy covariance, (4) hydrologic models, and (5) direct sap-flow measurements. The first three methods are integrated estimates for the entire vegetation–soil complex and provide estimates of evapotranspiration *not* transpiration. Hence, those methods do not directly partition water losses based on transpiration *versus* evaporation and provide no information on the source of water (i.e., shallow *versus* deep soil layers) for transpiration. Estimating transpiration with methods 1, 2, and 3 requires an independent analysis of the contribution of interception and soil surface evaporation. Hydrologic models vary considerably in complexity, ranging from very simple models [e.g., Thornthwaite (4) indices of potential evapotranspiration] to detailed physiologically based models that link vegetation, soils, and the

atmosphere (3). In contrast, sap-flow measurements provide a direct measure of transpiration (after correcting for time lags) under field conditions at the individual tree level (5–7). However, modeling or other scaling approaches are required to extrapolate tree-level measurements to the stand.

In summary, there are numerous approaches to quantifying transpiration in native or plantation-derived vegetative ecosystems. However, these methods vary considerably in accuracy, in data and measurement requirements, and in the capability to predict future transpiration rates as stands develop. In this chapter, we review approaches to quantifying forest transpiration from the leaf level to the stand and discuss the pros and cons of different approaches. We then provide applications of a subset of these approaches from phytoremediation case studies in Texas, Colorado, and Florida.

OVERVIEW OF CONTROLS ON TRANSPIRATION

Transpiration rates vary considerably among species and geographic regions (Fig. 1). Which factors contribute to this variation? At large scales (i.e., regions), climate is an overriding control. The strong relationship between evapotranspiration and precipitation (Fig. 1) suggests that transpiration is principally limited by soil water supply. However, other climatic factors such as temperature, atmospheric vapor pressure deficit, and solar radiation also play important roles and interact with soil water availability and physiological status of the plants (Fig. 2). For example, one of the key effects of temperature is through the influence on the length of growing season, in which longer periods with temperatures above freezing promote longer leaf area duration and hence, surface area available for transpiration. Frozen or cold soils also restrict transpiration (9) by limiting the permeability of cell membranes (10,11). Solar radiation provides the

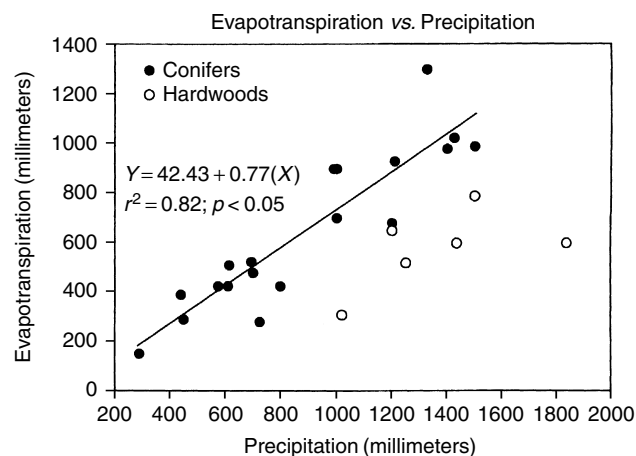


Figure 1. Relationship between annual evapotranspiration (Y) and precipitation (X) for hardwood and conifer species [redrawn from (3,8) and data from this chapter]. The regression line represents the data for the conifer species only. Note that r^2 is the correlation coefficient and p is probability.

energy for transpiration and regulates stomatal opening. As a result, a strong relationship generally occurs between solar radiation and transpiration, estimated as sap flow in Fig. 2. Atmospheric vapor pressure deficit provides the gradient to which leaf-water vapor responds through the leaf stomata (Fig. 2), and wind speed has a direct influence on the leaf boundary layer (12). Optimal climatic conditions for transpiration include high soil water availability, high solar radiation, high vapor pressure deficits, warm temperatures for extended periods, and high wind speed. In most cases, these conditions do not occur simultaneously because increased soil water availability is usually a result of high rainfall that decreases solar radiation (due to increased cloud cover) and vapor pressure deficit (due to higher humidity). Species that have the ability to utilize deeper sources of soil or groundwater [i.e., phreatophytic vegetation such as poplar (*Populus* spp.) and willow (*Salix* spp.)] are an especially attractive option in hot, dry, and windy environments

in the southwestern U.S., because transpired water can be derived from groundwater (13,14). Several studies have evaluated the influence of phreatophytes on surface and groundwater [e.g., (15–17)] from the perspective of negative impacts on streamflow and groundwater recharge. From a phytoremediation standpoint however, the high water consumption of phreatophytes has a positive effect to decrease aquifer recharge and influence the movement of contaminated shallow groundwater.

The structure, morphology, and physiological characteristics of the vegetation are also important regulators of transpiration. For example, at equal precipitation inputs, there are large differences in transpiration between conifer and hardwood species (Fig. 1), with hardwoods generally lower than conifers. Causes for these coarse scale differences are generally well known. The single greatest controlling factor is the quantity of leaves, expressed as leaf area index (in square meters per square meter). Site water availability and leaf area are related in that, sites with the

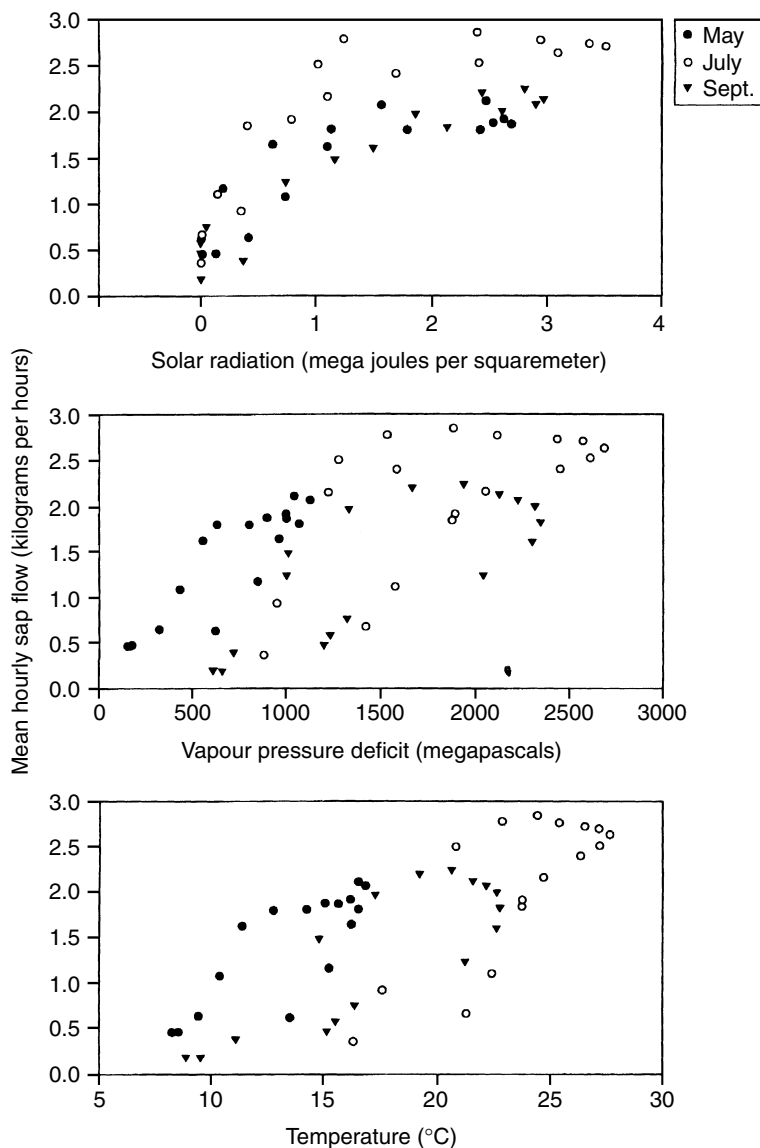


Figure 2. Mean hourly sap flows *versus* climatic driving variables for three seasonal measurement (May, July, and September) periods in central Colorado.

greatest water availability typically have the highest leaf area index (18,19), although nutrient availability (20,21) and temperature also play a role (22,23) in determining the maximum leaf area index. Watershed studies have documented strong relationships between leaf area and streamflow, with streamflow increasing exponentially as leaf area decreases (24,25). Because precipitation minus streamflow is an estimate of evapotranspiration at watershed scales, the implication is a direct control of stand-level transpiration by leaf area. Other structural and physiological factors regulating transpiration include the amount and permeability of sapwood and stomatal characteristics such as conductance and responsiveness to climatic variation and overall plant water status (i.e., water potential of leaves). Differences among species in leaf area, the rate of attainment of maximum leaf area, and physiological characteristics regulating the rate of water movement through the plant (sapwood amount and permeability, and stomatal conductance) provide opportunities for manipulating vegetation composition and structure to optimize transpiration. Optimal structural and physiological conditions for high transpiration amounts include rapid development of high leaf area, high stomatal conductance and sapwood permeability, and physiological characteristics that facilitate rapid responses to climatic conditions promoting transpiration.

Because transpiration is a function of root uptake from the soil and groundwater, soil characteristics are an important factor determining transpiration. Root growth and volume of soil occupied by roots are also important because water movement is slow when soils are drier than field capacity. Several factors determine soil water availability. First, the amount of precipitation entering the soil is a function of infiltration rate. Soils with low infiltration rates due to factors such as compaction or fine texture will have lower soil water availability because some precipitation may move across the soil surface in overland flow. Once in the soil, soil water availability is a function of water holding capacity and unsaturated hydraulic conductivity, both of which are determined by soil texture. Texture impacts water availability in different ways. Heavy clay soils (e.g., pore size less than 0.2 micrometer) have limited soil water availability because of very low rates of movement in the soil (i.e., conductivity) due to the fine pore space. In contrast, coarse textured sandy soils (e.g., pore size greater than 50 micrometers) have low water availability because of rapid drainage. Rooting volume and the presence or absence of restrictive layers are also important soil factors determining transpiration. For example, compacted soils provide a physical barrier to root growth, limiting root extension (26).

QUANTIFYING TRANSPIRATION

Leaf Level

Because water exits the plant primarily through leaf stomata (a small amount of cuticular transpiration may also occur in stems of some species), leaf-water relations are a key factor determining whole-plant transpiration (27).

The concentration gradient of water vapor between the interior of the leaf and the atmosphere at the leaf boundary layer defines the maximum transpiration rate. Vapor exchange is also determined by the opening size of the stomata. When stomata are wide open, transpiration occurs at about 20 percent to 40 percent of the rate of evaporation of open water (28), whereas closed stomata limit transpiration to less than 1 percent of open water. Stomatal opening is controlled by guard cell turgor, which responds to light, temperature, vapor pressure, and water potential of the leaves. The rate of movement of water through the stomata is the stomatal conductance. The rate of stomatal response to climatic conditions varies by species, but generally reflects responses to current conditions, whereas stomatal responses to water potential in leaves may reflect previous climatic and environmental conditions.

Because of the tight linkage between transpiration, leaf stomatal conductance (hereafter referred to as leaf conductance), and water potential of leaves, knowledge of all three parameters is useful for evaluating transpiration capacity. For example, species that exhibit high leaf-level transpiration and conductance, and maintain high water potential in the leaves have the capacity to transpire large quantities of water. Similarly, the relationship between water potential in the leaves and conductance is often threshold dependent; i.e., species that maintain high leaf conductance at low water potential have the capacity to transpire more water under dry conditions (29). Because of the importance of factors such as leaf area index and distribution, sapwood amount and permeability, and the difficulty in extrapolating spatially and temporally from the leaf to stand level, there may be no direct correspondence between leaf-level transpiration and overall stand transpiration. Typically, leaf conductance and tree and stand-level transpiration are most highly related in young stands with simple canopy architecture (7), such as closely spaced, even-aged monocultures. However, as stands develop, the linkage between leaf conductance and tree or stand-level sap flow declines due to shifts in the importance of stomatal *versus* boundary layer conductance to total vapor phase conductance (30,31). Hence, leaf-level measurements should only be used as an indicator of transpiration capacity.

Tree Level

Transpiration at the whole-tree level represents the integrated movement of water vapor from all the leaves in the crown of the tree. As mentioned in the previous section, spatial and temporal variation severely limits extrapolation of individual leaf measurements to the tree, so more direct measurements at the tree level are required. Two approaches have typically been used. In a few instances, entire trees have been enclosed in a cuvette and the flux of water vapor calculated based on the rate of increase in humidity within the enclosure. This approach is severely limited by methodological constraints such as the size of trees, heat buildup within the cuvette, and alterations in the boundary layer and vapor pressure gradients.

Sap-flow rate and volume have also been used as an estimate of transpiration (32). Because of lags between water movement in the stem and leaf-level transpiration, sap flow is not a direct measure of transpiration, but can be corrected after accounting for lags (33,34). Typically, a 1- to 2-hour lag correction is applied to real time sap-flow data to account for this temporal difference (7,34).

Two sap-flow techniques have been utilized; heat balance and heat pulse. For the heat-balance approach, collars consisting of a heating element and thermocouples above and below the heating element are placed around the stem and the entire stem section is heated. Sap flow is calculated using the heat-balance principle based on the difference in temperature between thermocouples above and below the heated stem section, after subtracting for heat loss due to conduction by stemwood (35). An advantage of this approach is that it integrates sap flow along the entire stem and does not require an independent estimate of sapwood area. For larger trees, paired probes are inserted vertically into the sapwood (36). The upper probe is heated and both contain thermocouples. The probes measure heat dissipation, which increases with sap flow and the resultant cooling of the heat source, as the apparent thermal conductance of sapwood increases with sap velocity. To convert sap velocity to sap flow rate, the cross-sectional area of sapwood must also be determined. Typically, trees are cored and sapwood to heartwood ratios quantified. Because sap-flow probes measure sap flow velocity at only one location, multiple probes are required to adjust for the variation in sapwood thickness and permeability in the stem section. Despite this, unaccounted for variation in horizontal and vertical variation in sapwood thickness and permeability introduces some error into sap-flow estimates obtained with probes. The magnitude of error can be determined experimentally and corrected for in small trees by comparing sap flow with actual transpiration using procedures such as weighing lysimeters. In large trees, corrections are much more difficult and hence, predictions have more uncertainty. In contrast to the heat-balance method, the heat pulse method estimates sap flow based on the time lag between pulses of heat and the distance between the sensors (37).

Stand Level

While it is informative to understand transpiration at the leaf and tree level to help evaluate species and environments suitable for phytoremediation, stand-level transpiration ultimately determines how much soil water and groundwater are removed. However, unlike leaf and tree measurements, no methods directly measure stand transpiration. Instead, three indirect measurement approaches have been utilized. These approaches involve gaged watersheds, extrapolation of individual tree measurements, and eddy flux estimates. Gaged watersheds require a combination of well-defined watershed boundaries, tight bedrock, and well-constructed weirs or gages to provide accurate transpiration estimates. If these criteria are met, then evapotranspiration (ET) is

estimated by the equation

$$ET = P - RO \pm \text{soil water storage} \quad (1)$$

where P = precipitation and RO = runoff, determined from weirs or gages. Because P is a component of the equation, the accuracy of precipitation measurements will also influence evapotranspiration estimates. Changes in soil water storage are usually assumed to be negligible at annual time steps, although this is clearly not the case over shorter intervals. Hence, using this approach at time steps less than a year requires determining changes in soil water storage. Because evapotranspiration is estimated, interception evaporation must be determined and subtracted to estimate transpiration.

Extrapolating individual tree measurements to the stand can be done in a number of ways. For example, instruments that measure sap flow can be installed on trees representing the averaged sized tree and mean sap flow multiplied times the number of trees in the stand (i.e., a "mean-tree" approach). Considerable uncertainty in stand-level estimates can accompany this approach where sites are variable. Alternatively, relationships between tree diameter, sapwood area, or basal area and sap flow at the individual tree level can be applied to all trees. In both approaches, repeated sampling is required to account for seasonal variability.

The eddy flux method uses water vapor gradients at fixed intervals above and below the canopy to calculate evapotranspiration. The technique is based on the assumption that water vapor flux is proportional to the vertical gradient of water vapor between two measurement points (averaged over several minutes). Typically, measurements are conducted from towers extending through the canopy. To be useful for estimating transpiration of a particular stand, the stand must be large enough to encompass most of the footprint measured by the sensors. In many phytoremediation applications conceived as of 2003, the stands are too small for an eddy flux approach to be appropriate.

Modeling

The use of modeling provides a potentially powerful tool for predicting current transpiration of native or planted vegetation and for projecting future transpiration capacity as a function of stand development. At the coarsest level of forecasting, gross measures of plant water demand and use can be derived from empirical estimates of potential evapotranspiration (4,38). These approaches usually consider climate and soils to some extent, but do not consider vegetation effects such as leaf area index, rooting depth, or leaf-level physiological characteristics. Hence, empirical approaches are useful for gross estimates of transpiration, but have limited utility for evaluating actual effects on the groundwater. At the other extreme, detailed physiological models that link the soil-plant-atmosphere continuum provide much more accurate estimates of transpiration. Depending upon the structure, models may also provide estimates of specific uptake locations within the soil profile (3,39,40). Using detailed physiologically

based models results in significantly greater data requirements. The most accurate application of these models requires site-specific estimates of soils, climate, and physiological characteristics of the major species on the site. However, large-scale application of detailed models with generalized parameters may provide estimates sufficiently accurate to be used in evaluating phytoremediation applications.

MEASURING AND MODELING TRANSPIRATION: CASE STUDY APPLICATIONS

Study Site Descriptions

Sap flow was measured at sites in Texas, Florida, and Colorado as components of larger studies evaluating the efficacy of using phytoremediation technology to clean up shallow groundwater contaminants. The north-central Texas study site was located, about 15 kilometers west of Fort Worth. The climate of this area is characterized as subhumid, with mild winters and hot, humid summers. The average annual precipitation is 80 centimeters per year with most rainfall occurring between May and October. Average annual temperature is 18.6°C. Study plots were located on the U.S. Naval Air Station, which adjoins U.S. Air Force Plant 4. A plume containing trichloroethylene was detected in the terrace alluvial aquifer in 1985. To demonstrate phytoremediation potential, eastern cottonwood (*Populus deltoides* Marsh.) trees were planted in two plantations over the TCE plume. One plantation was planted with vegetative cuttings (whips) and the other with 1-year-old nursery grown seedlings. Each plantation was approximately 80 by 20 meters and located perpendicular to groundwater flow in the alluvial aquifer. Sap-flow measurements were conducted using the heat-balance method (collars) in the first and second year after plantation establishment.

The eastern Florida site was located in the city of Orlando. The climate of the area is humid, with mild winters and hot, humid summers. The average annual temperature is 22.6°C and the average annual rainfall is 123 centimeters. Native vegetation of interest was located on the U.S. Naval Training Center. Trichloroethylene and tetrachloroethylene, which originated from a dry-cleaning facility that is no longer in operation, contaminate shallow groundwater. The plume extends under a 2-hectare forest and seepage wetland before reaching Lake Druid that borders the forest. A dense and diverse mix of overstory and understory species occur in the forest (density of 107 trees per hectare), with red bay [*Persea borbonia* (L.) Spreng.], camphor [*Cinnamomum camphora* (L.) Nees & Eberm.], slash pine and longleaf pine (*Pinus* spp.), sweet bay (*Magnolia virginiana* L.), and live oak and laurel oak (*Quercus* spp.) most abundant in the overstory. The most abundant understory species are skunk vine (*Paederia foetida* L.), saw palmetto [*Serenoa repens* (Bartr.) Small], cinnamon fern (*Osmunda cinnamomea* L.), and Christmas fern [*Polystichum acrostichoides* (Michx.) Schott.].

The central Colorado site is located approximately 20 kilometers southwest of Denver. The climate of the area

is dry, with warm summers and cold winters. Annual precipitation averages approximately 44 centimeters, with 30 percent of this amount received in April and May. The average annual temperature is 12°C. Study plots were located on the U.S. Air Force Plant PJKS. Trichloroethylene and dichloroethylene from a variety of sources contaminate the site. Measurements were conducted in two existing stands of natural vegetation: cottonwood–willow (*Populus* spp.–*Salix* spp.) and Gambel oak (*Quercus gambelii* Nutt.) The cottonwood–willow (*Populus* spp.–*Salix* spp.) stand is restricted to riparian areas (approximately 1 percent of the total land area of the site), while the Gambel oak (*Quercus gambelii*) stand is on more midslope locations (approximately 30 percent of the total land area of the site).

Methods

The sampling approach and methods varied among the three studies based on study objectives, species composition, and tree sizes. For the Texas study, sap flow from saplings in the plantation was estimated using sap-flow gauges (Dynamax Inc., Houston, TX) on 14 to 16 trees (divided equally among whips and 1-year-old trees) in May, June, July, August, and October over a 2-year period. During each measurement period, sap-flow measurements were taken every minute for 2 to 3 consecutive days. Data presented in this chapter represent averages of both plantations. In addition, sap flow was measured on nine larger native trees growing near the plantations using thermal dissipation probes (Dynamax, Inc., Houston, TX). Species sampled were: eastern cottonwood (*Populus deltoides* Marsh.), American elm (*Ulmus americana* L.), black willow (*Salix nigra* Marsh.), sugarberry [or large hackberry, (*Celtis laevigata* Willd.)], Eastern red cedar (*Juniperus virginiana* L.), and mesquite (*Prosopis pubescens* Benth.). At the end of sampling, increment cores were taken from the nine large trees for determining sapwood area.

For the Orlando study, sap flow was estimated using thermal dissipation probes installed on nine trees representative of major canopy species. Species sampled were: slash pine (*Pinus elliotii* Engelm.), longleaf pine (*Pinus palustris* Mill.), live oak (*Quercus virginiana* Mill.), laurel oak (*Quercus hemisphaerica* Bartram ex. Willd.), sweet bay (*Magnolia virginiana* L.), and camphor [*Cinnamomum camphora* (L.) Nees & Eberm.]. Two probe sets were installed into the sapwood on the north and south sides of sample trees, and sampling was conducted in November, March, and July for 2 to 3 consecutive days over a 1-year period. At the end of sampling, increment cores were taken and sapwood area determined.

For the Colorado study, sap flow was estimated using thermal dissipation probes on eight trees representing three species: eastern cottonwood (*Populus deltoides* Marsh.), narrow-leaf cottonwood (*Populus angustifolia* James.), and Gambel oak (*Quercus gambelii* Nutt.). Two probe sets were installed into the sapwood on the north and south side of sample trees and sampling was conducted in May, July, and September over a 1-year-period. At the end

of sampling, increment cores were collected and sapwood area determined.

For all three studies, data were summarized to provide average hourly sap flow rates (kilograms per hour) or daily totals (kilograms per day). In addition, climate was measured at all three studies with climate stations located on-site. Measurements included: hourly rainfall (centimeters), wind speed (meters per second), solar radiation (Watts per square meter), temperature ($^{\circ}\text{C}$), and relative humidity (percentage). Relative humidity and air temperature were used to calculate vapor pressure deficit (megapascals).

For the Texas plantation site, we parameterized and applied a mechanistic model of sap flow (PROSPER) and compared the results to sap flow measurements. Evapotranspiration at the Texas site was simulated because data were available to parameterize the model (7). The PROSPER model has been described in detail elsewhere (39,41), so only a general description is provided here. The PROSPER code is a phenomenological, one-dimensional model that links the atmosphere, vegetation, and soils. Plant and soil characteristics are combined into a single evapotranspiration surface that is characterized by a resistance to water vapor loss. This resistance is analogous to the relationship between stomatal resistance and water potential of the leaves and is a function of the water potential of the evapotranspiration surface. Evapotranspiration is predicted by a combined energy balance-aerodynamic method [Penman-Monteith equation modified as described in (42)] that is a function of the surface resistance to vapor loss described previously. The PROSPER model uses electrical network equations (41) to balance water allocation among vegetation and soil horizons. The flow of water within and between soil and plant is a function of soil hydraulic conductivity, soil water potential, root characteristics in each soil layer, and surface water potential. The PROSPER model predicts evapotranspiration, transpiration, and soil water distribution between soil layers daily, but monthly data are most accurate. The PROSPER model requires the following climatic data: solar radiation, precipitation, wind speed, air temperature, and vapor pressure. Initial model parameters include surface resistance to vapor loss, leaf area index, root distribution and surface area, soil moisture release, and several other parameters listed in Goldstein et al. (41).

Transpiration Estimates

Maximum transpiration rates for the study sites indicate large variation in transpiration potentials among sites (Table 1). On a per tree basis, rates ranged from 8 to 120 kilograms per tree per day. Much of this variation was related to differences in tree size that reflects differences in leaf area and sapwood area. For example, when pooling the data across the sites, a significant proportion of the variation in transpiration rates among and within sites can be explained by tree diameter (Fig. 3). Larger trees typically have greater sapwood volume resulting in more water transporting vessels (angiosperms) and tracheids (conifers) for sap-flow movement in the stems. Because leaf area is also related to sapwood area, larger trees will typically have greater leaf area index; and hence, greater surface area for transpiration.

When sap-flow rates are adjusted based on sapwood area (i.e., kilograms per day per square meter of sapwood), the variation in transpiration reflects species related differences in physiology (leaf, stem, and root), leaf area to sapwood area ratios, and site-dependent factors such as soil water availability and climate driving variables. Because species composition varies among sites and physiological and physical factors influence transpiration simultaneously, these studies

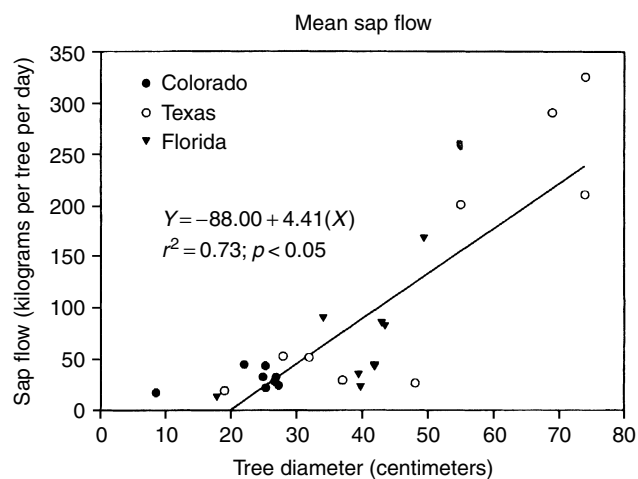


Figure 3. Total daily sap flows (Y) during peak transpiration periods (midsummer) *versus* tree diameter (X) across a range of species and site. Note that r^2 is the correlation coefficient and p is probability.

Table 1. Midsummer Peak Sap-Flow Rates Averaged Across Species and Measurement Days

Site	Sapwood Area (Square Centimeters Per Tree)	Sap-Flow Rates			
		Kilograms Per Day Per Square Meter of Sapwood	Kilograms Per Tree Per Day	Liters Per Hectare Per Year	Gallons Per Acre Per Year
Texas					
Plantation	30	2600	8	3 620 000	387 200
Native trees	820	1463	120	7 551 000	807 600
Colorado	234	1043	24	1 510 000	161 500
Florida	710	1535	109	6 859 000	733 600

cannot separate physiological and climatological effects; to do so requires an evaluation of transpiration rates of the same species and genotype in differing climatic and soil water availability conditions. For example, species sampled at the three sites represent a mixture of conifers, and ring porous and diffuse porous hardwood species, resulting in large differences in sapwood permeability and specific conductivity among sites and among species within sites (Fig. 4). In general, sap-flow velocity is lower in conifers and diffuse-porous species because sap flow moves through a number of annual rings, whereas water moves through only one or two annual rings in ring-porous species (11,43). Despite the limitations of the current approach, some notable patterns emerge when evaluating transpiration after adjusting for differences in sapwood area. For example, the cottonwood (*Populus deltoides*) plantation in Texas had the highest transpiration rate per unit of sapwood area, followed by the Florida stand, large trees in Texas, and the Colorado stand (Table 1). The high transpiration rate for cottonwood (*Populus deltoides*) in the plantation is a function of species characteristics that promote high transpiration, high leaf area per unit sapwood in the developing canopy, and access to shallow groundwater. In contrast, transpiration rates per unit of sapwood area were lowest in Colorado, even though the site contained cottonwood (*Populus deltoides*) and several of the measured trees occurred in the riparian zone. The combination of species composition and climate characteristics were not as conducive to high sap-flow rates per unit sapwood area relative to the other sites.

Using sap-flow techniques to predict actual stand transpiration requires frequent sampling to account for seasonal variation. Ideally, sap flow should be measured continuously for the entire growing season on a

large number of trees. Because this approach is often impractical, an alternative is to measure sap flow at shorter frequencies and calculate bounds or maximum values as a tool to evaluate phytoremediation potentials. Because the sampling frequency varied among the case studies described here, we focused only on measurements during the highest transpiration period (midsummer). These estimates can be used as a “best-case scenario” approach—that is, if these rates occurred on the site, would transpiration be sufficient to control the plume? To estimate maximum potential transpiration at the stand level (i.e., kilograms per hectare or gallons per acre), we extrapolated the tree transpiration data (kilogram per tree per day) assuming a 180 days transpiration period and a stem density of 350 stems per hectare, except for the plantation where actual tree density was used (Table 1). We emphasize that these data provide estimates of maximum transpiration capacity under *in situ* climate conditions because the peak sap-flow rates were used in the extrapolation and previous studies have shown considerable seasonal variation in sap flow (7). The 350 stems per hectare is representative of a fully stocked stand under most forest conditions and is consistent with full canopy closure and maximum leaf area index.

When comparing results from the sites with mature trees, the variation in maximum transpiration capacity is considerable. The Texas site has a maximum transpiration capacity of approximately 7.5×10^6 liters of water per hectare per year (8×10^5 gallons of water per acre per year) if the site was fully stocked with the sampled species. By contrast, the Colorado site has a maximum transpiration capacity of approximately 1.5×10^6 liters per hectare per year (1.6×10^5 gallons per acre per year). The plantation site in Texas currently has a maximum transpiration capacity of approximately 3.7×10^6 liters per hectare per year (4×10^5 gallons per acre per year). However, we anticipate that transpiration will equal or exceed the estimate from mature trees on the site (*versus* 7.5×10^6 liters per hectare per year or 8.0×10^5 gallons per acre per year) once the canopy develops and achieves the maximum leaf area.

Comparison of Measured versus Modeled Transpiration

A critical need for phytoremediation is the development and application of a tool to provide species and site-based estimates of transpiration. While a powerful tool for measuring transpiration from vegetation already on-site or quantifying transpiration of planted vegetation, sap-flow measurements at every phytoremediation site may not be practical. One potential tool for application across sites is the development or application of models. In most cases, however, models need to be calibrated or parameterized for specific site and species conditions. To evaluate the use of such a tool, we parameterized PROSPER for the Texas plantation using intensive site (soils, climate, and root distribution) and leaf-level measurements (stomatal characteristics and leaf area index) [see (7)]. We compared monthly transpiration estimates obtained with PROSPER to transpiration estimated from sap-flow measurements

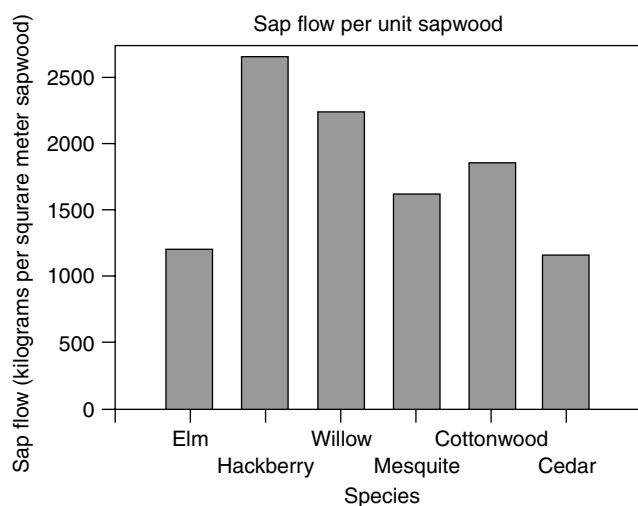


Figure 4. Mean growing season sap-flow rate per unit sapwood for six species in north-central Texas. Black willow (*Salix nigra* March.), eastern cottonwood (*Populus deltoides* Marsh.), and eastern red cedar (*Juniperus virginiana*, L.) are diffuse porous species, while American elm (*Ulmus americana* L.), hackberry (*Celtis laevigata* Willd.), and mesquite (*Prosopis pubescens* Benth.) are ring porous.

over a 2-year period (Fig. 5). Comparisons indicated generally good agreement between predicted and measured values, except during the late summer that was coincident with some of the driest and hottest periods (August and September 1998) in our study. During this period, PROSPER predicted a considerable decline in transpiration, while measured values showed an increase. We attribute this discrepancy to an inability of PROSPER to adequately simulate root uptake from shallow groundwater during drought conditions, since the original formulation of PROSPER was designed to only simulate surface and soil water dynamics (39,41). The results of this comparison are consistent with other studies that have shown that PROSPER provides reasonable estimates of either evapotranspiration or transpiration (3,40). However, refinements in the subsurface water and groundwater hydrology and subsequent availability to tree roots might improve the predictive capability and usefulness as a phytoremediation evaluation tool.

CONCLUSIONS AND RECOMMENDATIONS

The importance of transpiration to the success of phytoremediation applications suggests that accurate estimates of current and potential transpiration should be a high priority when considering this approach for site management. Both measuring and modeling transpiration are important. Assessments require a substantial sampling commitment for direct measurements or parameterizing physiologically based models. This requires detailed knowledge of local site conditions and physiological parameters for the major species. For screening assessments, we recommend that published estimates be used to set the bounds for maximum transpiration capacity based on general climate and vegetation characteristics of the location. If these general transpiration rates are great enough to influence groundwater hydrology, then evaluations of current, enhanced (e.g., manipulating the structure and species composition of current vegetation), or new vegetation transpiration capacity should proceed. Technology

and models exist to provide reasonable estimates and predictions of transpiration. However, the accuracy of the estimates depends on the investment in accounting for the spatial and temporal variation or in providing site and species-specific estimates for physiologically based transpiration models.

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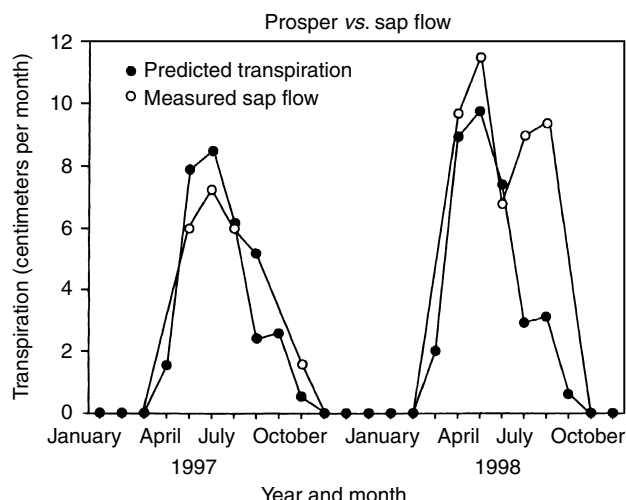


Figure 5. Comparison of measured sap flow and predicted transpiration from PROSPER for cottonwood (*Populus deltoides* Marsh.) plantations in north-central Texas.

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WATER LOGGING: TOPOGRAPHIC AND AGRICULTURAL IMPACTS

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Water logging occurs on poorly drained soils. Under periods of heavy precipitation or snowmelt, water percolates very slowly, and soils become wet in short

periods of time. Some soils have inherently low ability to infiltrate and hydraulic conductivity, especially heavier clay soils. Soil compaction because of heavy farm equipment further reduces bulk density, which in turn reduces the ability to infiltrate soil. The result is that the water table accretes and may eventually rise to the land surface. High-intensity rainfalls also lead to ponding of water on the soil surface. These conditions of surface ponding, a saturated soil profile, and gradual rise of the water table to the soil surface all lead to water logging. One very specific hydrogeologic condition that contributes to water logging is groundwater artesian pressure.

Land topographic features have an influence on water logging. On very flat lands, with low hydraulic gradients to rivers and watercourses, surface runoff is not easily drained. Water is also easily trapped in surface depressions on undulating lands. Low-lying coastal and riverine areas are very susceptible to flooding in the wet rainy periods. Towns, villages, and agricultural lands suffer from backwater effects. Infrastructural works sometimes exacerbates water logging. For example, during heavy rainfalls, water becomes trapped behind road and railway embankments. The situation could be extremely damaging under monsoon conditions in Asia, where widespread flooding occurs with loss of life and property.

Water logging also occurs in the arid and semiarid regions on poorly drained soils, because of excess irrigation water applications and canal seepage. It is estimated that about 25–30 million hectares of irrigated land suffer from water logging.

Globally, some areas of permanently wet or water logged regions exist, which are known ecologically as wetlands, and they serve very important environmental and hydrologic functions. Wetlands provide habitats for waterfowl, flora, fish, and animals. They also replenish groundwater, provide flood control, and filter pollutants. Organic soil deposits, or peat bogs, are also characterized by naturally high water tables.

It is estimated that over 200 million hectares of agricultural land worldwide suffer from poor drainage. Crops need an optimum soil-air environment in the root zone in which to thrive. Under high water table conditions or poor drainage, crop growth is retarded because of a lack of oxygen in the root zone. Root and shoot growth are affected by high water table conditions, as are other plant physiological conditions. For example, stem elongation is slowed, senescence and abscission of older leaves occur, rapid stomatal closure exists, and leaves begin to yellow. Crop failure is inevitable under prolonged flooding. Anaerobic conditions lead to reduced oxygen, and an accumulation of carbon dioxide, ethylene, other toxic substances, and byproducts in the soil profile, all of which are harmful to plant growth.

Some plants have the ability to withstand high water tables better than others. For example, rice is grown under flooded conditions. Some berry crops, e.g., highbush blueberries and cranberries grow under saturated soil conditions. Native vegetation and trees can be found in swamps, marshlands, and peat bogs. Crop tolerance to high water tables not only depends on crop type, but also on stage of crop growth. For example, most crops are

likely to be killed by flooding during germination and early vegetative growth. As plants mature, they are slightly more tolerant to high water tables. However, vegetable crops are very sensitive to high water tables during their entire growth period. Deeper rooted crops, such as grasses, sugarcane, corn, shrubs, and trees, are somewhat hardier.

Apart from rice and some berry crops, it is impossible to undertake large-scale commercial plant and animal production without removal of excess soil water. Drainage is therefore essential for lowering the water table on poorly drained soils. The most common form of drainage is surface drainage. Open ditches are dug in fields to evacuate excess water. Lands are also shaped either by forming bedded fields or by a ridge and furrow system. In this way, crops can be grown on the raised beds or ridges. Other forms of land shaping include land leveling and smoothing, or grading the fields with a slope toward open drains. Surface depressions in fields can also be filled in by land leveling, and this will reduce ponded water on the land surface. Tile drainage, or subsurface pipe drainage, is another form of drainage for improving the productivity of poorly drained agricultural lands.

Tile drainage or tubewells are also used to lower the water table in irrigated lands. If seepage from irrigation canals is determined to be a major contributor to the water logging of adjacent lands, the canals can be lined, or buried pipe interceptor drains are installed adjacent to the canal. Relief wells are normally installed in lands affected by artesian uplift.

In the lowing coastal and riverine areas, flood control structures are built, which include levees, dykes, walls, and berms. Sluice gates or pumping stations are constructed to evacuate water from behind these structures to either the sea or rivers.

WEED CONTROL STRATEGIES

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The control of nuisance vegetation in watercourses is often necessary to ensure adequate flood defense for surrounding land, to provide facilities for recreational sporting activity, to provide better angling environments, to aid navigation, on public health grounds, for industrial uses, and to maintain adequate water supply and quality. In recent years, weed control for conservation has also become necessary to protect native habitats from invasion by alien, introduced, or nonnative species.

Aquatic weeds are different from terrestrial weeds in that they are usually a normal constituent of the aquatic ecosystem. Aquatic plants become weeds when they grow to excess and become a nuisance by blocking channels, reducing flow, or preventing other normal uses of the waterbody. Increased urbanization has resulted in an increase in the number of nonnative species now considered aquatic weeds, due primarily to escape of plants

from domestic situations. The increased recreational pressure on enclosed inland waterways has also resulted in greater demands for aquatic plant management.

During the last 50 years, there has been a dramatic change in the type of aquatic weed problem. Emergent weeds, such as reeds and rushes, and floating leaved weeds, such as waterlilies, were the major problem, but eutrophication since the Second World War, as a result of intensification of agriculture, increased urbanization, and consequent sewage treatment requirements, has resulted in a shift to problems caused by filamentous and blue-green (cyanobacterial) algal blooms.

TYPES OF AQUATIC WEEDS

Although there are a large number of individual aquatic weed species, for the purposes of control, they can be divided into four major groups.

Emergent Plants

Emergent plants are rooted in the sediment at the margins of watercourses. They have stems and leaves that protrude above the water's surface. Growth is often limited by increasing water depth, making this type of weed common along the edges of water. Some common characteristics of these plants are long narrow leaves, and heights between 1 and 3 m. Species include *Phragmites communis* (common reed), *Typha latifolia* (bulrush), *Schoenoplectus lacustris* (common club rush), *Glyceria maxima* (reed sweet grass) and *Sparganium erectum* (common bur reed).

Smaller emergent and bankside plants, except grasses, tend to be broad-leaved weeds. These include *Rorippa nasturtium-aquaticum* (watercress) *Apium nodiflorum* (fool's watercress), *Berula erecta* (water parsnip), and others.

Other plants in this group are predominantly submerged but may form different types of leaves when they become emergent due to lowered water levels or grow above the water surface. These include some *Ranunculus* spp. (water crowfoots) and *Hippuris vulgaris* (mare's tail).

Plants that grow on banks at or above the water line should be included in this group. Common weeds in this class include *Lythrum salicaria* (purple loosestrife), *Epilobium hirsutum* (greater willow herb) and *Phalaris arundinacea* (reed canary grass).

A number of riparian weeds associated with watercourses can be included in this group. The dispersal of these species is furthered by flowing water. The alien invasive species *Fallopia japonica* (Japanese knotweed), *Heracleum mantegazzianum* (giant hogweed) and *Impatiens glandulifera* (Himalayan balsam) are included here.

Floating Plants

Free-Floating Plants. This group includes many of the worst types of aquatic weeds, such as *Azolla* spp. (water fern), *Eichhornia crassipes* (water hyacinth), *Lemna* spp. (duckweeds), *Pistia stratiotes* (water lettuce) and *Salvinia molesta* (giant water fern), all of which have been found in Britain. They are characterized primarily by very rapid vegetative reproduction and spread.

Floating-Leaved Plants. This type of plant is characterized by rooting in the sediment and forming floating leaves. They may be rooted at the margins and form floating mats over the surface of the water, such as *Hydrocotyle ranunculoides* (floating pennywort). This species has leaves held above the water surface. Plants rooted in deep water that have floating leaves held flat on the surface include *Potamogeton natans* (broad-leaved pondweed), *Nuphar lutea* (yellow water lily) and *Nymphaea alba* (white water lily).

Other plants in this group are predominantly submerged, but may form different types of leaves when they reach the water surface, such as the rosette type of leaf produced by *Callitriche* spp. (water starworts).

Submerged Plants

This group includes all submerged plants except algae. It can be divided into two sub-groups, those rooted in the sediment, such as *Elodea* spp. (waterweeds), *Myriophyllum spicatum* (spiked water milfoil), and *Potamogeton pectinatus* (fennel pondweed), or those free-floating below the water, such as *Ceratophyllum demersum* (rigid hornwort) and *Lemna trisulca* (ivy-leaved duckweed).

Algae

Algae can be divided into two types for the purpose of control, macrophytic types, including filamentous algae and charophytes, and unicellular types, including pea-soup algae and cyanobacteria or blue-green algae. Nuisance algae tend to grow best in nutrient-rich still or slow flowing water.

THE BIOLOGY AND ECOLOGY OF AQUATIC WEEDS

Seasonal Growth and Dispersal

Aquatic plants are usually a problem only during the summer months. Various forms of dieback or washout during autumn and winter ensure that very little aquatic vegetation is left to overwinter. However, nuisance aquatic weeds use several strategies to survive winter. These include seed production by annual riparian weeds and as an insurance policy by perennial submerged aquatic macrophytes. Others produce turions (specialized leaf buds) tolerant of anoxic conditions, for example, *P. pectinatus*, fennel pondweed. The majority of emergent macrophytes have dormant roots and rhizomes. Algae use spore production and *Azolla* and others can overwinter as intact plants, for example, *Callitriche* spp. and *Crassula helmsii*.

Aquatic weed growth usually starts when mean diel water temperature rises above 6°C; this is normally a week to a few weeks behind terrestrial plant growth. However, growth is rapid once started and stem elongation in *Ranunculus* spp. can produce 6 m of growth before flowering. Shortening day length usually initiates flowering so the majority of aquatic plants flower in or after the last week in June in the temperate Northern Hemisphere.

Stem fragmentation and vegetative reproduction is the most common form of dispersal within systems, and seed

production is not considered important for maintaining weed populations. However, establishment of new weed populations is most likely to derive from establishment of seeds. Birds, animals, machinery, boats, and man can also spread viable fragments of weeds to new sites.

Characteristics of Nuisance Plants

Growth Rate. Species that have extremely fast growth rates tend to become greater problems than other species. For example, duckweed and *Azolla* can double their biomass in less than 4 days in good conditions.

Morphology. Species that produce dense masses of vegetation impede the flow of water. Others have rigid stems that may impede boat traffic, interfere with access to the water, and trap detritus, forming temporary dams that may cause flooding.

Dispersal Strategies. Plants that produce new plants by fragmentation are also very effective aquatic weeds. Often, mechanical control is responsible for increases in the distribution of these species in watercourses.

Toxicity and Taint. Some plants are toxic to animals, and others taint potable waters. Blue-green algae produce toxins that can kill fish, and other animals and also impart an unpleasant taste to drinking water.

CONTROL METHODS

Factors Affecting Choice

A number of factors influence the choice of weed management technique, including the cost of control. These factors are related to the uses of the water and the specific landscape situation of the waterbody. Often, a compromise between weed control efficacy and other considerations must be made.

Type of Control. In some cases, it is not desirable to attempt complete eradication of the weed population, and localized control is preferable. Selective control can be either by type of species or by, for example, clearing swims for anglers. Total control may be required where flood defense is an issue or where nonindigenous species are present. Temporary alleviation of a weed problem is sometimes possible by frequent mechanical control during the season.

Use of the Water. Certain weed control practices may not be compatible with the use of the water. The most obvious of these is when water is abstracted for drinking water supplies downstream of an intended herbicide application site. The limit of $0.1 \mu\text{g L}^{-1}$ for individual pesticides set by the EU Drinking Water Directive imposes strict requirements on the distance between the site of application and the abstraction point. Dilution and flow rate affect this distance.

Human and Livestock Safety. This includes direct toxicity of herbicides to operators and water users,

increases in turbidity due to stirring up of sediments, and unpleasant odors from rotting vegetation or herbicide residues in water. The safety of farm and other animals can be compromised by the direct toxicity of the herbicide, or, as is more likely, by increasing the palatability of treated poisonous plants.

Irrigation. The hazards to adjacent crops should be considered through irrigation with water containing pesticides or spray drift at the time of application and the spread of weeds such as *P. australis* into cropland when dredged spoil is deposited onto land adjacent to watercourses.

Industry. Weeds blocking intake sluices or the presence of pesticides in water may affect industrial processes requiring water.

Environmental Impacts. In general, the more effective a weed clearance operation, the greater the environmental impact. Deoxygenation of water, which affects fish, can occur when large volumes of cut or herbicide-treated weed are allowed to rot *in situ*. The number of invertebrates removed by mechanical control is estimated at approximately 1 million per m^3 of cut weed, and the number of vertebrates, mainly small fish, is estimated at about 40 per m^3 of weed removed. Care should be taken to avoid removing marginal vegetation during bird nesting times. Other indirect effects include destabilization of banks by removal of plant cover or by deposition of thick layers of cut vegetation on sloping banks.

MANAGEMENT OPTIONS

Mechanical Control

The following points should be considered when using mechanical weed control. Rafts of cut weed can drift downstream and block sluices, pumps, weirs, and other water control structures. Deoxygenation can occur if weed is left to rot in water, posing a risk to fish and invertebrate life. Rapid regrowth of cut weeds often necessitates a late season cut. In practice, the cost of mechanical control limits the frequency of the operation, thus a compromise between weed control and cost is usually necessary.

Cutting. Aquatic weeds can be cut manually or by using machines; the choice of technique depends on the scale of the problem to be tackled. Cutting by hand and raking cut material onto banks is not used widely now, although the majority of small drainage ditches used to be managed in this way. Now a wide range of mechanical cutting boats and weed bucket attachments for bank-based machines is available, but the principles of cutting and removing the cut weed remain unchanged. In general, the deeper the cut, the longer the weed control will last. Timing of the cut is critical. Good control can be achieved by regular cutting throughout the growing season, but this may not be possible for operational reasons. Cutting submerged weeds before the end of June will require a further cut toward the end of the season, and regrowth during summer tends to

be faster than initial growth rates. The timing of cutting emergent and bankside weeds is less critical, but care should be taken to avoid bird nesting seasons.

Weed boats are used to cut submerged weeds in relatively deep and wide watercourses where large areas of weed require cutting. They usually have a front mounted reciprocating cutting bar which can cut to a depth of about 1.5 m. The work rate is estimated at approximately 1 km per day on a 10-m wide river. Cut weed must be removed from the water, either by a separate boat or by the action of a front rake on the cutting boat after cutting.

Bank-mounted equipment includes draglines, flail mowers, and dredging equipment. The work rate for cutting buckets and weed rakes is, on average, about 500 m day⁻¹ on wide rivers.

Harvesting. The use of aquatic weed harvesters combines the cutting and collecting operation, therefore reducing time and increasing efficiency. They can and/or collect all types of weed, including filamentous algae and duckweeds and can deposit them at a collection point on the bank. The work rates for this type of equipment vary considerably with the density of weed and the distance traveled to offload the collected weed.

Uses for Cut Weed. There are a limited number of uses for cut weed; these include composting, reed for thatching, mulches, soil conditioners, and cattle forage. The material has low value, and processing should be done locally to minimize costs.

Chemical Control

Herbicides can be used to control a wide range of aquatic plants. They may be applied to emergent or surface-floating weeds by a foliar spray in much the same way as recommended for land plants. Submerged weeds and algae are treated by adding the herbicide to the water to build up an effective concentration in the water. This method will also control some floating and emergent species. Chemicals currently used to control aquatic plants are listed in Table 1.

Consideration should be given to the effects of herbicide treatment on the function of the waterbody. This not only involves the direct toxicity of the chemical but also indirect effects caused by decomposition of dead weeds. Retaining a proportion of the weed growth is recommended to maintain a habitat for invertebrates, fish, and other wildlife and to stabilize the ecosystem. Vegetation left *in situ* will take up nutrients and help to prevent the growth of algae. To minimize the effects of decomposing weed, herbicide labels have recommendations on treatment intervals that should be observed. It is normal to treat only 25% of an infested waterbody at one time, separated by the application interval. Consideration should be given to the irrigation interval when using herbicides in water. This is the time between application and the time when water is safe to use for irrigating of crops or watering livestock.

Application Methods

Application to Foliage. Recommendations for the correct dose are made in the form of weight of product or active ingredient per hectare. Herbicides are usually applied by knapsack or boat-mounted sprayer equipment. Using high-volume and low-pressure equipment to deliver a coarse spray with a minimum of small droplets is advisable to reduce the risk of spray drift. Localized or selective control can be achieved by the appropriate choice of herbicides and by spraying only those plants required. Water velocity and quality do not affect the treatment, but to avoid a buildup of chemical in the water, it is normal to apply herbicides in an upstream direction.

Application to Water. Recommendations usually refer to the theoretical concentration of active ingredient that would be achieved when the chemical has been evenly distributed throughout the water body but before any adsorption or degradation has occurred. This is usually expressed as parts per million (ppm or mg l⁻¹ or g m⁻³). Some formulations may be applied on a rate per surface area basis because they sink onto the weeds or mud.

Table 1. Herbicides Suitable for Controlling Major Weed Groups

	Asulam	Copper	2,4-D amine	Dichlobenil	Diquat	Diquat Alginat	Endothall	Fluridone	Glyphosate	Maleic Hydrazide	Terbutryn
Trees and shrubs on banks									●		
Bracken and docks	●				○				●		
Broad-leaved weeds			●	○	○			○	●		
Grasses				○	○			●	●	●	
Reeds and sedges					○		●	●	●		
Floating-leaved plants				●	○	●	●	○	●		○
Free-floating plants			○		●			○	●		●
Submerged plants			○	●	●	●	●	●			●
Submerged plants (flowing water)						●	○				
Algae		●			●	●		○			●

● suitable for control; ○ short-term control or some species within groups not susceptible

The methods of application depend on the formulation. Herbicides should be spread as evenly as possible over the surface of the water. Granular formulations can be spread by hand using a suitable container, by a mechanical spreader, or by using an air-assisted blower. Liquid formulations should be diluted and applied by subsurface injection. This is usually achieved by trailing nozzles below the water as close to the top of the weed bed as possible, without disturbing the bottom sediment. Viscous gel formulations should be applied with specialized equipment.

Timing. Emergent and floating leaved plants should be treated from midsummer, when the leaves have formed fully. Annual weeds should be treated before flowers have set to prevent production of seeds.

Treatment of submerged weeds and algae is normally recommended in spring and early summer when weeds are actively growing. Treatment of floating-leaved plants is usually more successful before the floating leaves have formed. Later in the season, when large biomasses of weed have developed and particularly when the water is warm, there is a severe risk of deoxygenation of the water, after control, caused by decomposition of the dead weed. This effect is particularly acute when using terbutryn, as the mode of action blocks photosynthesis very quickly, but respiration continues.

An indirect hazard to fish may arise through possible effects on fauna that provide food for fish. These may be due to direct toxicity to invertebrates or more likely to loss of habitat after weed control operations. However, loss of habitat can lead to increased predation by fish. These effects are temporary and should be set against the disadvantages of dominant weed species reducing habitat structure for all aquatic fauna.

Biological Control

The decline in public acceptability of herbicide use in water, coupled with an increase in the costs of mechanical

control has led to the development of several biological control agents for aquatic weeds.

Livestock. Horses, cows, goats, and sheep can be used to graze marginal vegetation. Care should be taken to avoid excessive poaching of the banks (although some poached banks are a valuable habitat), and fencing is required to keep animals from straying.

Chinese Grass Carp and Other Fish. Herbivorous fish are most common in Asia and South America. One species, the Chinese grass carp, *Ctenopharyngodon idella*, is available for weed control in enclosed situations. Triploid grass carp are used in the United State to ensure that the fish cannot breed. Licenses are usually required to introduce of these fish. Stocking densities vary depending on the type and severity of the weed problem, but usually a density of between 75 and 150 kg ha⁻¹ will achieve adequate weed control. Bottom feeding fish stir up sediment and create turbid conditions in which submerged macrophytes cannot grow. These fish include primarily carp and bream.

Waterfowl. Waterfowl graze on some species of floating and submerged weed. When present in large numbers (e.g., swans on *Ranunculus*), they can cause significant losses of aquatic vegetation, which may not be desirable.

Pathogens and Insects. Classical biological control is not widely practiced in Europe, and not at all on aquatic and riparian weeds. Biological control agents survive only on their host species, meaning they cannot spread to ornamentals or related crop species. Most native plant species have a number of native host-specific pathogens or insect predators. There are effective biological control agents for many aquatic weed species (Table 2) established in other parts of the world, but none is used in Europe.

Biological control has been successful in about 45% of the attempts made on aquatic weeds. It is essentially a management tool designed to maintain the level of a pest

Table 2. Established Biological Control Agents for Aquatic Weeds

Weed Species	Biological Control Agent	Type	Region
<i>Alternanthera philoxeroides</i>	<i>Agasicles hygrophila</i>	Flea beetle	America
<i>Azolla filiculoides</i>	<i>Stenopelmus rufinasus</i>	Weevil	S. Africa
<i>Eichhornia crassipes</i>	<i>Neochetina eichhorniae</i>	Weevil	America Africa, India
	<i>Neochetina bruchi</i>	Weevil	America, Africa, India
	<i>Cercospora piapori</i>	Fungus	Africa
	<i>Altenaria eichhorniae</i>	Fungus	Africa
<i>Hydrilla verticillata</i>	<i>Hydrilla pakistanae</i>	Insect	America
	<i>Hydrellia sarahae</i>	Insect	
	<i>Hydrellia pakistanae</i>	Insect	
	<i>Hydrellia balciunasi</i>	Insect	
	<i>Mycleptodiscus terrestris</i>	Fungus	
	<i>Plectosporium tabacinum</i>	Fungus	
	<i>Cricotopus lebetis</i>	Midge	
<i>Myriophyllum aquaticum</i>	<i>Lysathia sp</i>	Beetle	S. Africa
<i>Myriophyllum spicatum</i>	<i>Bagous subvittatus</i>	Insect	N. America
	<i>Bagous myriophylli</i>	Insect	
	<i>Phytobius</i>	Insect	
<i>Salvinia molesta</i>	<i>Cyrtobagous salviniae</i>	Weevil	Africa
	<i>Neohydronomus affinis</i>	Weevil	

at a tolerable level in the community without necessarily eradicating the weed species. It is a useful tool to employ in combination with other techniques such as mechanical or chemical control.

Environmental Control

Environmental control is the technique of changing the environment, either temporarily or permanently, to reduce the suitability of the habitat for the target weed. This technique tends to shift the weed species to other plants, which may or may not be desirable depending on the ultimate use of the waterbody in question.

Flow. Aquatic plants are divided into riverine or lacustrine species. Although not mutually exclusive, plants that tend to favor slow flowing environments do not grow in faster flowing water and vice versa. Alterations to flow can have marked effects on the composition of the aquatic plant community.

Water Chemistry. Reduction of the quantity of major plant nutrients entering water tends to reduce the biomass development of weed species but not necessarily the species composition of the weed community. Diversion or further treatment of sewage effluent to remove excess P has been effective in stabilizing river communities by removing the dominance of filamentous algae. Nutrient release from sediments is a major source of supply for lakes and other slow flowing watercourses, such as canals and drainage ditches. Even if all nutrients could be prevented from entering such systems, there would be sufficient nutrient supply in the sediment to provide for continued aquatic plant growth for more than 100 years in most cases.

Shade. Shading submerged aquatic macrophytes can be an effective method of control, especially in sensitive areas where chemical control is inappropriate. Shade can be produced by planting trees or shrubs on the banks of watercourses. However, this often interferes with access to the water and may not be appropriate in all situations. Opaque floating material can be used in still waters to achieve increased shading, although it must remain in place for at least 4 months to have a lasting effect. The use of dyes that absorb light at photosynthetic wavelengths is also widely used in amenity lakes and ponds. Increased turbidity from bottom feeding fish also excludes light, as does the action of powered boats, and these techniques are most effective in sluggish water where the sediment can remain suspended for long periods.

Burning. Emergent weeds, particularly the stiffer stemmed reeds, do not always collapse in the autumn and can form large masses of dead, standing material. In dry ditches and on banks, this material can sometimes be burned. This can be a useful way of reducing the bulk of plant material which might otherwise collapse during the winter and block channels. Burning can also be used to destroy cut material after drying. This is good way of disposing of poisonous plants, some of

which remain toxic after death but become palatable to livestock.

Alterations in Water Level. Exposure of part, or all, of the bed of a water body by lowering the water level has been used successfully to manage aquatic vegetation. Control is achieved either by dehydration of the vegetation or by exposure to low temperatures. The process is sometimes termed drawdown or dewatering. It can also alter the character of the sediment which may reduce weed growth. However, in deeper water bodies, drawdown can allow weeds to establish in depths below their normal limit. Once established, they can grow up toward the surface, as the level is again raised, to remain in the higher light intensity. In these situations, drawdown can spread a weed problem into areas which would normally remain weed-free.

Some indication of the relative effectiveness of treatments can be seen in Table 3. This is not a definitive guide for every situation, and expert advice should be sought before applying any technique to a particular weed problem to optimize efficacy and minimize costs.

The benefits assigned to each technique are for general guidance only, and the results obtained may be modified by specific site conditions.

Other Methods of Control

Many other techniques for controlling aquatic vegetation have been tried, which do not fit closely into any of the previous categories. These include the following: (1) the use of lasers to control emergent or floating weeds and ultrasound vibrations that can disrupt cells of submerged plants; (2) floating oil films that cause floating weeds to sink, and (3) increasing wave action that submerges floating weeds and increases turbidity thus suppressing submerged vegetation. There are also several biological control agents, including crayfish, other invertebrates, and wildfowl, that have been effective against some weeds. Most of these have been tested in tropical or subtropical conditions and, at present, the grass carp is the most effective biological control agent in more temperate regions. Some of the more effective alternative treatments are outlined below.

Magnetic Treatment of Flowing Water. The use of magnetic water treatment devices, it has been shown, affects the growth of algae. This is thought to be due to interference with the uptake and storage of calcium in algal cells. This treatment is most effective in closed recirculating systems.

Ultrasound. Ultrasonic devices are used widely in Europe to disrupt cells mainly of algae and biofilm bacteria in water storage reservoirs where herbicides cannot be used.

Miscellaneous Treatments. There are a number of alternative treatments for water that affect the nutrient concentration and, therefore availability, to aquatic plants. Most of these involve precipitation of phosphate in an insoluble form.

Table 3. The Relative Efficacy of Management Options

Management Option	Weed Type					
	Emergent		Floating-leaved		Submerged	
	Narrow-leaved	Broad-leaved	Rooted	Free-floating	Rooted	Algae
<i>Mechanical</i>						
Hand-cutting	○	○	○	✗	○	✗
Flail mower	○	○	✗	✗	✗	✗
Weed bucket	○	○	○	○	○	○
Weed boat	○	○	○	○	○	✗
Harvester	○	○	○	○	○	○
Dredger	✓	✓	✓	✗	✓	○
<i>Chemical</i>						
Foliar application	✓	✓	✓	○	✗	✗
Applied to water	✗	✗	✓	✓	✓	✓
<i>Biological</i>						
Livestock	○	○	✗	○	○	✗
Waterfowl	✗	✗	✗	✓	○	○
Fish	✗	✗	✓	✓	✓	✓
Insects and pathogens	○	○	○	✓	✓	○
Biomanipulation	✗	✗	✗	✗	✗	✓
<i>Environmental</i>						
Shade	○	○	○	○	○	○
Nutrient manipulation	✗	✗	✗	○	○	○

✓ Effective control lasting at least one season

○ Moderate benefit only, or control lasting less than one season

✗ No useful effect

SCREEN FILTERS FOR MICROIRRIGATION

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Water quality is a major concern in the management of microirrigation systems (drip, microsprinkler, microspray). Emitters plugged by physical, chemical, or biological contaminants may create significant problems in everyday maintenance. Effective and reliable filtration is required for successful operation of microirrigation systems. There are different types of filters available for the removal of physical contaminants from irrigation water. Selection of a filter depends on the type and amount of contaminants in the irrigation water, the type of emitters, the size of the irrigation system, and the desired management practices. Screen filters should be a primary choice when water is pumped from a well where the only filtration requirement is to remove mineral particulate matter (Fig. 1). They are sufficient in the absence of organic contamination and in addition they are usually inexpensive and easy to maintain. To remove both particulate matter and organic growths, media filters (also called sand filters) may also be used. They are usually recommended when large amounts of algae or other organic contaminants are present.

Screen filters are simple and economical filtration devices of various shapes and sizes that are frequently

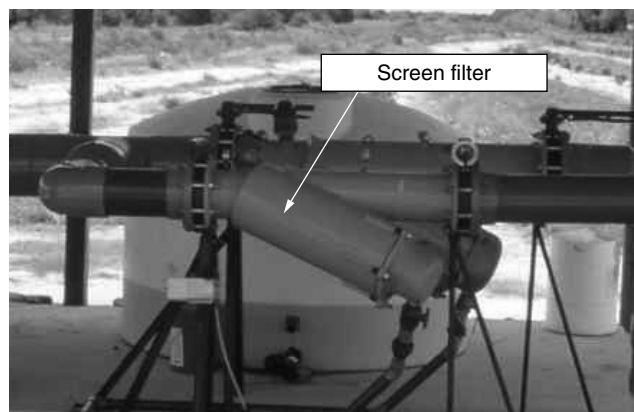


Figure 1. Screen filter installed in the main line of irrigation system.

used for the removal of physical contaminants. They can be made of metal, plastic, or synthetic cloth enclosed in a special housing. Screen filters are recommended for the removal of very fine sand or larger-sized inorganic debris from irrigation water. It is normally not effective to use screen filters for the removal of heavy loads of algae or other organic material typical in surface water supply, since screens clog rapidly, requiring too frequent cleaning to be practical.

When surface water is used for microirrigation, screens are often used as secondary filters, after organic matter has been removed with media (sand) filters. In this

capacity they prevent washed out media from entering the irrigation system. When well water is used for irrigation, a screen filter may be used as a primary filter, or it may be secondary to a vortex sand separator, depending on the mineral particle load in the water.

Filtering screens are classified according to the number of openings per inch, with a standard wire size given for each screen size (Table 1). The minimum size of particle retained by a screen filter with a certain mesh can be determined from Table 1. A 200-mesh screen is usually recommended for drip type microirrigation where 150 mesh is often sufficient for microsprinklers and microsprays. Water filtered with a 200-mesh screen will contain only smaller particles of very fine sand, silt, and clay. These particles should freely move through the emitter openings unless flocculation, aggregation, and bridging are taking place due to unusual water chemistry. If clogging problems persist with 200-mesh filtration, chemical treatment of water may be necessary.

Generally, it is recommended to remove particles down to a size ten times smaller than the emitter's passageway so that grouping and bridging of particles will not cause clogging. The maximum particle size that can safely enter the irrigation system with a given emitting device should be provided by the manufacturer. Organic particles with a density approaching the density of water tend to group and bridge more easily. Particles heavier than water, typically mineral particles, may settle and collect in the low flow zones of the irrigation system.

The size of the filter is specified by its effective area, which is the area of the openings in the screen. It is specified in relation to cross-sectional area of the main pipe. A desirable ratio is 2 or more (area of openings much larger than the cross-sectional area of the pipe). The mesh size of the filter (opening size) will depend on the smallest particle size to be removed from the irrigation water.

Table 1. Representative Screen Mesh Numbers and the Corresponding Standard Opening Size Equivalents

Screen Filters		Removed Particles	
Screen Size, mesh	Opening Size, μm	Soil Classification	Particle Size, μm
4	4760	Very coarse sand	1000–2000
10	2000		
20	711	Coarse and medium sand	250–1000
40	420		
80	180	Fine sand	100–250
100	152		
120	125		
150	105		
180	89	Very fine sand	50–100
200	74		
270	53		
325	44	Silt	2–50
— ^a	— ^a	Clay	2 and less

^aScreen filters are not normally used to remove this size particles.

When estimating the appropriate size of filter for a specific application, one should consider the quality of water needed, volume of water required to be passed through the filter between consecutive cleanings, filtration area of the filter screen, and allowable pressure drop through the filter. A screen filter can handle a large range of discharges. However, discharges that are large in relation to the filtering surfaces will result in greater pressure losses, shorter life of the filter, and the requirement for frequent cleaning.

If the irrigation water contains large amounts of sand or other inorganic contaminants, it is advisable to use vortex sand separators before the screen filters, which use centrifugal force to separate heavier particles away from the fine mesh cartridge. Heavy particles collect at the bottom of the filter, where they can be periodically flushed away.

Simple gravity screen filters are often used as primary filtration where an elevated water source from a canal or reservoir is available. Since the pressure loss through a gravity screen filter is very small, these filters are often used in systems where such losses must be minimized. They operate on the principle of gravity force, not water pressure. Because only atmospheric pressure is used, soft organic material is not forced through the screen. Gravity screens are easy to install and maintain. Usually, these filters are constructed as two chambers separated by a fine mesh screen, which can be cleaned manually or automatically. Most of them incorporate self-cleaning water jets.

If water is pumped from a surface source (lake, pond, stream, river, irrigation canal, or pit), aquatic plants, fish, and animals should be kept away from the intake pipe since the efficiency of pumping may be reduced by debris in the water entering the suction pipe. This can be accomplished with an intake screen. The design of the screen must allow for the separation of the debris from the water without pulling the debris against the screen. A horizontal screen placed below the water surface or an intake basket-type filter are often used for this purpose. Some of these screens are self-cleaning and require very little maintenance.

Small amounts of larger particles, such as lime rock flakes from the aquifer, can be removed using a Y-type strainer with replaceable screens and with clean-out faucets or valves. They can be flushed while in use if the required high flow for flushing is provided. Blowdown filters operate in the same way as other types of pressure screen filters. They are designed for easy cleaning, which is done by opening a valve that diverts the water flow through the screen, releasing trapped particles.

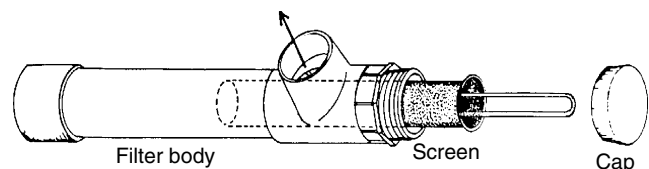


Figure 2. Schematics of a typical screen filter.

Occasionally, blowdown filters require manual cleaning to eliminate particles that have become lodged in the screen.

Screen filters can be cleaned manually by removing the screen and washing it with clean water (Fig. 2). However, dismantling is not very practical for use with waters containing high levels of contaminants, since it must be done very often. Blowdown screen filters greatly simplify the cleaning process. They are cleaned with the water diverted through the screen by opening a valve. This operation does not require disassembling. The screen does not have to be removed for routine cleaning of the filter, but it must be removed for occasional cleaning of the particles that become lodged in the screen.

The most convenient method for cleaning screen filters is automatic screen flushing triggered by critical pressure differential across the filter, which increases with buildup of debris on the screen. Automatic cleaning can also be set on a time schedule. Filtered water is always recommended for washing and backflushing. Additional safety devices such as secondary screen filters at the entrance to each irrigation system subunit are highly recommended.

XERISCAPE

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Much attention and controversy have surrounded the *xeriscape* concept of *landscaping* since its inception in 1981. The proper definition of xeriscape is "quality landscaping that conserves water and protects the environment." Above all things, it must be a quality design that balances the lawn area, shrub and flower plantings, and the hardscape (i.e., decks, patios and sidewalks). Landscapes composed of rocks or plastic flowers alone are not xeriscapes. Xeriscapes are in tune with the environment; therefore, xeriscape applies to the desert in southwestern United States as well as the semitropical southeast.

Xeriscape has seven basic principles:

- planning and design
- soil improvement
- appropriate plant selection
- practical turf areas
- efficient irrigation
- mulching
- appropriate maintenance

USE OF PRACTICAL TURF AREAS IN A XERISCAPE DESIGN

Of the seven principles, none has received more attention than practical turf areas. This principle, which concerns turfgrass in the landscape, has been shrouded in misinformation that has been touted as fact by "experts" in xeriscape, water supply, and turf culture.

The original turf-related principle established by the Denver originators of xeriscape was "limited turf use." For Denver and much of the arid West, the seemingly logical

approach to reducing landscape water consumption was simply to reduce the use of turf. However, as the Xeriscape concept has matured and spread, the principle of limited turf use was increasingly scrutinized by horticulturists and turf experts. Today's xeriscape movement incorporates a more holistic approach to reducing turf irrigation, fully recognizing that the type of plant materials or irrigation in the landscape has as much of an effect on water consumption as the human factor and good landscape water management (Fig. 1).

THE NEED TO CHANGE ATTITUDES AND HABITS

Throughout the xeriscape movement, the evident truth is that plants do not waste water; people do. Another fact is that irrigation systems do not waste or save water; people do. The mission of xeriscape is clear: Change the attitudes and irrigation habits of professional and amateur landscape managers. Proper water management provides the greatest opportunity for *water conservation* in the landscape.

Xeriscape focuses on the use of turfgrass in the landscape because of the tremendous potential for irrigation water abuses in the name of maintaining green turfgrass. Within the traditional landscape, turfgrass has received the major share of total landscape irrigation because grass often makes up a large percentage of the total landscape. Through the principles of xeriscape, turf irrigation can be reduced and the many benefits of turfgrass can still be derived.

BENEFITS OF TURFGRASS IN THE LANDSCAPE

Turfgrass is an integral component of most landscapes. It is certainly the best recreational surface for children and athletes. Furthermore, it has a tremendous mitigating effect on the environment, reducing heat loads, noise and water and air pollution. A turfgrass lawn is second only to a virgin forest in the ability to harvest water and recharge groundwater resources. As a design component, turfgrass provides the landscape with unity and simplicity while inviting participation in it.

However, the fact remains that turfgrass is the highest user of irrigation water in the traditional landscape. This



Figure 1. The proper definition of xeriscape is "quality landscaping that conserves water and protects the environment." Above all things, it must be a quality design that balances the lawn area, shrub and flower plantings and the hardscape (i.e., decks, patios and sidewalks).

is significantly different from saying that turfgrass is the highest water-using plant in the landscape—which is not the case. The discrepancy between these two statements yields the most common misconception and misrepresentation in xeriscape, and it is therefore the basis of controversy and unproductive efforts. To resolve this controversy, some scientific and practical fundamentals of turfgrass are explained using actual xeriscape principles.

XERISCAPE PRINCIPLES FOR REDUCING TURFGRASS IRRIGATION

Specifically, xeriscape principles promote the following strategies to reduce turfgrass irrigation:

- Prepare soils for turf areas as carefully as any other planting area to use all the moisture available, promoting the plant's vigor and water-use efficiency.
- Place turf species in landscape zones based on water requirements.
- Select adapted turf species and varieties that have lower water demands.
- Irrigate turf in areas that provide function (i.e., recreational, aesthetic, foot traffic, dust and noise abatement, glare reduction, temperature mitigation).
- Use nonirrigated turf areas where appropriate.
- Irrigate turf based on true water needs.
- Decrease fertilization rates and properly schedule fertilization.

FINE-TUNING TURFGRASS XERISCAPE PRINCIPLES

In traditional landscape design, turfgrass makes up the major portion of landscapes. The tremendous square footage of turfgrass in a landscape accounts for the reason that turfgrass irrigation, as a percentage of total landscape irrigation, is so high. The “practical turf areas” guideline promotes the use of turf only in those areas of the landscape that provide function. In residential landscapes, a turf area is usually a necessity for recreation and entertainment. But turf should not be irrigated on narrow strips of land or other areas that are difficult to water (Fig. 2).

Good landscape water management begins with planning and design. By designing the landscape as zones based on plant water needs, turf can be appropriately placed for function, benefit and water efficiency. Zoning the landscape and irrigation system allows watering turfgrass on a more frequent schedule than shrubs. For established trees and shrubs, the irrigation strategy should use deep soil moisture and depend on natural rainfall to replenish soil moisture. When sufficient rainfall does not occur, supplemental irrigation of trees and shrubs may be required.

Another way to incorporate turf into the landscape and conserve water is simply not to irrigate. Many turfgrass species are drought-tolerant and can survive extreme drought conditions. The grass may turn brown for a while, but rainfall will green it up again. This approach may be unacceptable for many residential and commercial



Figure 2. In residential landscapes, turf areas are important for recreation and entertainment.

landscapes, but in the case of parklands, industrial sites and rights-of-way, brown turf may be acceptable.

SELECTING THE PROPER TURFGRASS SPECIES AND VARIETIES

Wherever the landscape, selection of turfgrass species and varieties is of utmost importance. Extensive research has shown that there are significant differences in water requirements among turf species and even among varieties within species. The capacity of different turf species to avoid and resist drought also varies significantly. To help reduce landscape water requirements, xeriscape recommends selecting turfgrass varieties (and other landscape plants) that are both adapted to the area and have the lowest practical water requirements.

Landscape managers should be keenly aware of drought-stress indicators shown by turfgrass and other plants in the landscape, including a range of color changes, leaf curl, and wilting, and they should strive

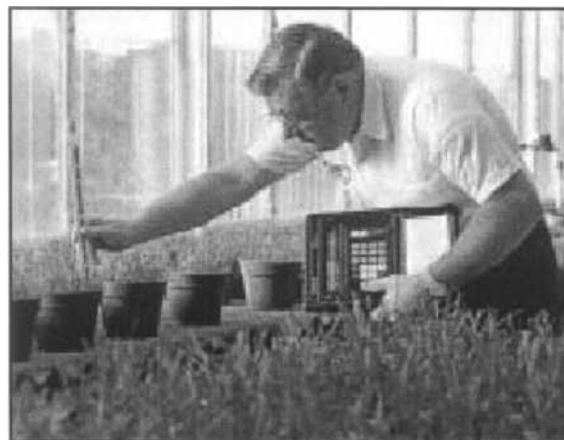


Figure 3. Wherever the landscape, selection of turfgrass species and varieties is of utmost importance. Extensive research has shown that there are significant differences in water requirements among turf species and even among varieties within species.



Figure 4. Attitudes and habits about turf are changing. For much of the arid West, the seemingly logical approach to reducing landscape water consumption was simply to reduce the use of turf. However, as the xeriscape concept has matured and spread, the principle of limited turf use was increasingly scrutinized by horticulturists and turf experts. Today's xeriscape movement incorporates a more holistic approach to reducing turf irrigation, fully recognizing that the type of plant materials or irrigation in the landscape has as much of an effect on water consumption as the human factor and good landscape water management.

to meet the water needs of each group of plants. By irrigating only when the plants require water versus by the calendar, the manager can dramatically reduce landscape water use.

The water requirements of turfgrasses can be minimized through specific horticultural practices. Decreasing fertilizer application rates and timely applications of slow-release fertilizers tend to reduce flushes of growth that can increase water requirements (Fig. 3).

THE XERISCAPE CHALLENGE

Xeriscape is a challenge and an opportunity for the "green" (landscape, turf and nursery) and "blue" (water utilities and agencies) industries. Through xeriscape, these two industries have been brought together to focus on landscape water use. Although this marriage has not always been easy, the best minds are prevailing in efforts to perfect and implement the xeriscape concept.

By embracing the xeriscape concept, including the principle of practical turf areas, the green and blue industries can continue to be recognized as good stewards of the environment (Fig. 4).

MEDIA FILTERS FOR MICROIRRIGATION

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To avoid plugging of microirrigation emitters, it is generally recommended to remove particles down to a size ten times smaller than the emitter's passageway so that grouping and bridging of particles will not cause clogging. The manufacturer should provide the maximum particle size that can safely enter the irrigation system with a given emitting device. Organic particles with a density approaching the density of water tend to group and bridge more easily.

Media (also called "sand") filters are well suited for removal of either organic or inorganic particles. Due to their three-dimensional nature, media filters have the ability to entrap large amounts of contaminants. They do not seal off as easily and therefore will not clog as often as filters that trap particles on their surface, such as screen filters.

Media filters used in microirrigation systems operate under pressure. They consist of fine gravel and "sand" (that actually is crushed granite or silica) of selected sizes placed in pressurized tanks (Fig. 1). The main body of the tank contains sand, which is the active filtering ingredient. The sand is placed on top of a thin layer of gravel that separates it from an outlet screen. Sharp-edged sand or crushed rocks are recommended for the filtering media since the sharp edges catch soft algal tissue. Crushed granite or silica graded into specific sizes for a particular system are commonly used porous media. It is not recommended to use natural sand as a filtering media since the particles are usually rounded and smooth. The size of media particles is very important. Too coarse particles result in poor filtration and possible clogging of the emitting devices, while too fine particles trigger unnecessary frequent backwashing of the filter.

Two factors describe the media used in the filter: uniformity coefficient and mean effective size. Uniformity coefficient reflects the range of sand sizes within the grade. It is desirable to keep sand particles as uniform in size as possible. The uniform size of filtering media assures better control of the filtration. Grading in size from fine to coarse causes premature clogging of the filter, since the nonuniform pores of the media also retain particles small enough to be tolerated by the emitters. The uniformity coefficient is represented by the ratio of the size of the screen opening that would pass 60% of the filter sand to the screen opening that will pass 10% of the same sand. For irrigation purposes a uniformity coefficient of 1.5 is considered adequate.

The mean effective sand size is the size of the screen opening that will pass 10% of the sand sample. It is a measure of the minimum sand size in the grade and therefore an indicator of the particle size that will be removed by the media. Some examples of sand media are shown in Table 1. It can also be seen that the quality

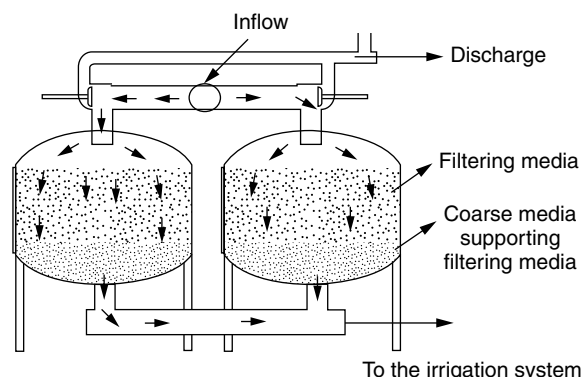


Figure 1. Schematic of media filters in filtration mode.

Table 1. Relationship Between Designated Media Numbers, Filtration Quality, and Equivalent Screen Sizes

Corresponding Screen Size, mesh	Opening Size, μm	Removed Particles		Media Filters		Filtration Quality, mesh
		Soil Classification	Particle Size, μm	Media Number	Material	
4	4760	Very coarse	1000–2000	8	granite	100–140
10	2000	sand				
20	711	Coarse and	250–1000			
40	420	medium sand				
80	180	Fine sand	100–250			
100	152					
120	125			11	Granite	140–200
150	105			16	Silica	140–200
180	89	Very fine	50–100	20	Silica	200–230
200	74	sand				
270	53			30	Silica	230–400
325	44	Silt	2–50			
— ^a	— ^a	Clay	2 and less			

^aScreens are not normally used to remove this size particles.

**Figure 2.** Media filters installed in a field.

of filtration increases with a smaller effective size of filtering media.

The design flow capacity of a media filter is expressed in liters per second of flow per square meter of surface area of the media normal to the direction of flow. The filter should be sized so it can handle the poorest water quality at a given site and provide the required flow for the functioning of the irrigation system (Fig. 2). The quality of irrigation water before filtration may vary with the seasons and weather conditions. This is especially true with regard to surface water but can also apply to well water in some cases.

The effectiveness of a filter is a measure of its ability to remove particles of a certain size. The effectiveness of filtration increases with a decreasing size of filtering media grain size (larger designation number). However, smaller media size will require more frequent cleaning.

Filtration effectiveness is also inversely proportional to the flow rate through the filter. The higher the flow rate, the lower the effectiveness of the media filters. Therefore, filter manufacturer's specifications and water quality samples should be used to select a filter for a specific application.

Media filters are cleaned by backwashing (Fig. 3). This operation consists of reversing the direction of water flow in the tank. Clean water is usually supplied from the second tank. The upward flow fluidizes the media and flushes collected contaminants. The backflush water is discharged and does not enter the irrigation system. The backwash flow must be carefully adjusted in order to provide sufficient cleaning without accidental removal of the media. If flows larger than those recommended are used to backflush the media, it is very likely that the media will be flushed from the filter by the backflow water. Only recommended flow rates for a given media should be used.

Most media filters are backflushed at prescheduled time intervals or by using automatic devices based on the pressure loss across the filter. For systems in which low

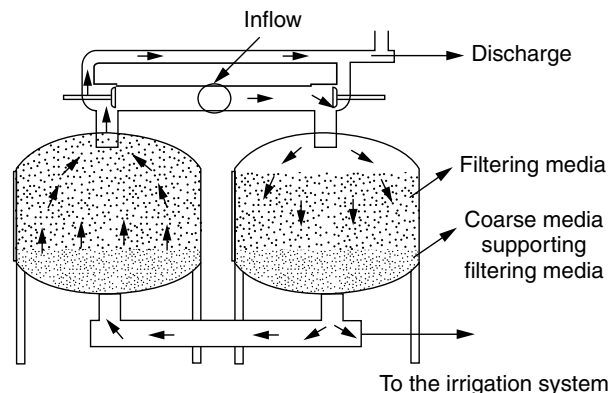


Figure 3. Schematic of the media filters in backflush mode.

quality water requires frequent backflushing, automatic cleaning is necessary to avoid problems.

As the filter is backflushed every time the pressure differential exceeds a predetermined value, large pressure drops in the irrigation system are avoided, maintaining the system's uniformity and efficiency. The pressure differential triggering backflushing depends on the pressure required for the proper functioning of the irrigation system. The irrigation system pump should be able to supply enough pressure to compensate for the pressure drop through the filter just before the filter is washed. Pump capacity must also be adequate to supply enough pressure and flow rate to flush the filter. Also, it is advisable in a large irrigation system to use a number of smaller tanks instead of a few large ones because of the higher backwash flow that large tanks require for good cleaning. Automatic backflush systems will eliminate sudden changes in water quality that can create problems if a filter is washed only at regular intervals. This is especially important in systems using surface water supplies with changing contamination levels.

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